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Recommendations Concerning Models and Parameters Best Suited to Breeder Reactor Environmental Radiological Assessments

Charles W. Miller
Charles F. Baes III
Donald E. Dunning, Jr.
Elizabeth L. Etnier
Kathy K. Kanak
David C. Kocher
Craig A. Little
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RECOMMENDATIONS CONCERNING MODELS AND PARAMETERS BEST SUITED TO BREEDER REACTOR ENVIRONMENTAL RADIOLOGICAL ASSESSMENTS

Charles W. Miller   David C. Kocher
Charles F. Baes III[1]   Craig A. Little
Donald E. Dunning, Jr.   Laura M. McDowell-Boyer
Elizabeth L. Etnier   H. Robert Meyer
Kathy K. Kanak   Elizabeth M. Rupp
                Roberta W. Shor

[1]Environmental Sciences Division

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HIGHLIGHTS

A project is underway at Oak Ridge National Laboratory which has as its objective the identification of models available for environmental radiological assessments; evaluation of model structure, simplifying assumptions and data bases; estimation of their uncertainties; and, if possible, the recommendation of the models and parameters which are best suited to particular assessment situations. When needs are identified, recommendations are also made for further environmental and biomedical research.

The purpose of this report is to present recommendations concerning the models and parameters best suited for assessing the impact of radionuclide releases to the environment by breeder reactor facilities. These recommendations are based on the model and parameter evaluations performed during this project to date. Seven different areas are covered in separate sections of the report. These sections may be summarized as follows:

The Gaussian plume model continues to be appropriate for most estimates of atmospheric dispersion when the dispersion parameters which are used account for the release height and terrain conditions under consideration. Ground deposition and plume depletion processes can be parameterized through the use of the deposition velocity.

The terrestrial food chain model assumes equilibrium conditions and requires the input of empirically derived transfer coefficients. Wherever possible this report describes the statistical distribution of each of these transfer coefficients for a given radionuclide.

A two dimensional, K-theory model assuming no sorption of the radionuclide[1,2] seems appropriate for many hydrologic transport calculations.

If sorption and sediment transport are important considerations, however, more complicated models[34] are needed to estimate the transport in surface water systems.

For assessment purposes, the bioaccumulation factor may be used to relate the concentration of a radionuclide in water to its concentration in aquatic foodstuffs. Bioaccumulation factors associated with 32 elements are tabulated for marine and freshwater fish and invertebrates.

The internal radiation dose to man depends in part on the quantity of radionuclides incorporated into the body from food, water and/or air, and hence intake rates of these materials are important. Adult intake factors are presented for fruits, vegetables, grains, meats, fish, poultry, milk, other liquids, and air.

Fifty-year dose commitment factors from inhalation and ingestion of potentially significant radionuclides in breeder reactor fuel cycles are presented for lungs, total body, ovaries, total endosteal cells, and testes. These calculations are based on the latest criteria provided by the International Commission on Radiological Protection and other recognized authorities. Also tabulated are β and photon external dose conversion factors for body surface, lungs, ovaries, skeleton, testes, and total body due to immersion in contaminated air and exposure to contaminated surfaces.

The methodologies currently used at ORNL for estimating the dose to man from airborne releases of ^{3}H and ^{14}C are delineated. However, these methodologies are presently undergoing a critical review as part of another project at ORNL. The methodologies outlined in this report may be modified upon completion of that review.

Work on the model evaluation project is continuing at this time. As this work proceeds, modification of the recommendations presented in this report may be made to reflect the latest findings of the project staff.

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1. INTRODUCTION

Charles W. Miller

Currently, a number of environmental transport and dose calculation models employing different data bases and assumptions are available for the assessment of the radiological impact of routine and accidental discharges. The model evaluation project initiated at Oak Ridge National Laboratory (ORNL) has had as its objectives identification of these models; evaluation of their structure, simplifying assumptions and data bases; estimation of their uncertainties; and, if possible, recommendation of the models and parameters which are best suited to particular assessment situations. Where needs are identified, recommendations are also made for further environmental and biomedical research. The models examined in this project to date include those developed for the prediction of atmospheric and hydrologic transport and deposition, terrestrial and aquatic food-chain bioaccumulation, and internal and external dosimetry.

The purpose of this particular report is to make recommendations concerning models and parameters for use in assessing the impact of breeder reactor radioactive discharges to the environment. These recommendations are based on the results of the model evaluations project to date. It should be noted, however, that this project is still underway, and these recommendations may be adjusted as new information is analyzed.

This report contains summaries of the recommendations for each of the transport and dosimetry areas listed above. These summaries reflect the work of those individuals who have contributed significantly to the

project. Further details concerning the models and parameter values discussed may be found in the literature cited.

For assessment purposes, the simplest model which can be acceptably validated is deemed the most suitable for a given set of conditions.[1] The models considered in this report are based heavily on the recommendations made by the model evaluation workshop[2] held as part of this project. Additional analyses of model uncertainty were also used in selecting models for inclusion in this report.[3]

Whenever possible, the input parameter values used in assessment models should be based on site-specific information. This, however, is often not practical. In the absence of site-specific information, default values must be chosen for each model input parameter. Often, only a single value is chosen for each parameter of interest. In reality, however, each input parameter has a distribution of values associated with it.[1]

During the course of the model evaluation project, data for a number of parameters have been examined in an attempt to estimate the distribution of values associated with a given parameter. However, it was generally found that few of the parameters analyzed have a data base large enough to represent the true distribution. The assumption that the available data are representative of the true population of the parameter value of interest is also crucial as the data considered do not necessarily represent unbiased samples.[4]

In this report, information on the estimated distribution associated with a given parameter value is presented wherever possible. If the data for a given parameter were lognormally distributed, the data were

log-transformed to produce a normal distribution, and estimates of the population mean μ and standard deviation σ of the logarithms were determined. If the data were found to be normally distributed, the population mean $\overline{X}$ and standard deviation (S.D.) were estimated. If this information is available for a given parameter, the cumulative probability associated with any value chosen for the parameter can be determined.[1,4] The values of the various parameters to be used in an assessment calculation will depend on the cumulative probability desired in the final answer. Therefore, recommendation of a single value for any parameter for use in assessment calculations biases model predictions. Before choosing a parameter value an assessor should consult the cited literature to determine the potential effect of the limitations noted above on the final results.

Such an in-depth analysis has not yet been performed for all parameters and all elements considered in this report. For these situations, the parameter values currently in use at ORNL have been entered and their origin cited. These values may change, however, as this project continues.

The dose conversion factors given in this report are based on model calculations rather than observations. It is not possible to validate these models because of the impracticality of measuring dose in human subjects. These models, however, are based on the best animal and human data currently available.

A separate section in this report is devoted to the dosimetry of ^3H and ^{14}C. These nuclides are often of special interest because of the importance of hydrogen and carbon in biological systems and because they are important effluents of various fuel cycles. Methodologies for assessing the dose to man from these nuclides are currently being evaluated

by another project at ORNL. This report presents the methodology currently used for assessment purposes at ORNL. This methodology may be modified, however, as the evaluation continues.

As mentioned previously, work on this project is continuing. Current efforts are focused in three major areas:

1. comparison of predictions from various environmental transport models with measured field data to estimate the uncertainty in model output;

2. continued determination of distributions associated with model input parameter values;

3. development of recommendations for needed environmental and biomedical research.

As this work continues it may result in modification of the information contained in this report.

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2. ATMOSPHERIC DISPERSION AND DEPOSITION

Charles W. Miller

One of the principal ways in which radionuclides from breeder reactor facilities reach the environment is through discharges to the atmosphere. These discharges result in doses to man as a result of:

1. exposure to the contaminated air; and

2. exposure to surfaces and ingestion of foodstuffs contaminated by radionuclides removed from the plume by deposition processes.

2.1 Atmospheric Dispersion

Atmospheric dispersion calculations provide estimates of air concentration resulting from atmospheric discharges of radionuclides. These air concentrations are then used to calculate doses to man.

2.1.1 The Gaussian plume model

The Gaussian plume model[1] is the most widely used method of estimating downwind air concentrations of radionuclides released to the atmosphere.[2,3] It is also the most often verified atmospheric dispersion model.[4] For a continuous point source and invariant meteorology, this model is given by

$$\chi = \frac{Q}{\pi u \sigma_y \sigma_z} \exp\left[- 1/2 \left(\frac{H}{\sigma_z}\right)^2 - 1/2 \left(\frac{Y}{\sigma_y}\right)^2 \right] \quad , \qquad (2.1)$$

where

χ = ground-level air concentration (Ci/m^3),

Q = release rate (Ci/sec),

H = height of release (m),

u = mean wind speed over the height of the mixing layer (m/sec),

σ_y, σ_z = standard deviation of a Gaussian distribution in the cross

wind y and vertical z directions, respectively (m).

For estimating the impact of routine (i.e., continuous) radionuclide releases, the average ground-level concentration in air over a sector 22.5° wide is often used. It is given by

$$\chi = \frac{2.032 \, Q}{x\bar{u}\sigma_z} \exp\left\{-1/2\left(\frac{H}{\sigma_z}\right)^2\right\} \quad , \qquad (2.2)$$

where

x = the downwind distance of interest, m;

$\bar{u}$ = average wind speed during time period of interest.

The Gaussian plume has a number of theoretical limitations.[5] However, when properly used it has been found to be a very practical tool for dispersion modeling. It is

1. mathematically simple and flexible,

2. in accord with much, but not all, diffusion theory, and

3. a reliable framework for the correlation of both field studies and mathematical and physical modeling studies of atmospheric diffusion.[5]

2.1.1.1 <u>Dispersion parameters</u>. It has been found that the dispersion parameters, s_y and s_z, need to be carefully specified when using the Gaussian plume model.[6,7] A number of empirically determined graphs of s_y and s_z as a function of downwind distance and atmospheric stability have been proposed.[8] Geiß et al.[9] have published a useful set of comparisons between some of these sets of graphs.

Whenever possible, site-specific measurements should be used to specify σ_y and σ_z.[10] When sets of standard curves are used instead, at least two such sets should be used, one for surface releases and one for elevated releases. The Pasquill-Gifford curves[1] adjusted for averaging time and surface roughness have been suggested for use with the former[10,11] (Table 2.1). For elevated releases, the curves suggested by Geiß et al. or those measured at Brookhaven National Laboratory[12] have been recommended[10,11] (Table 2.2).

As noted above and in Tables 2.1 and 2.2, σ_y and σ_z depend on atmospheric stability. A number of different methods for classifying stability have been proposed, but these methods often give significantly different results when applied to the same meteorological data set.[13,14] Until enough data become available to select the best method of classifying stability, the user of the Gaussian plume model must exercise care in choosing stabilities. The selection of a stability category alone can result in a factor of four difference between the lowest and highest annual average air concentration estimated from a given set of σ_y and σ_z curves.[13,14]

2.1.1.2 <u>Release height</u>. The effective release height, H, is also a critical parameter in Eqs. (1) and (2).[6,7] The value of H depends on more than just the physical height of the stack, h:[15]

$$H = h + h_{pr} - h_t - C \ , \tag{2.3}$$

where

h_{pr} = rise of plume above the release point due to buoyancy and momentum,

Table 2.1. Coefficients for specifying the Pasquill-Gifford[9]
system of dispersion parameters for six stability categories[9]

	Atmospheric stability category					
Coefficient[a]	A	B	C	D	E	F
a_1	-0.023	-0.015	-0.012	-0.0059	-0.0059	-0.0029
a_2	0.35	0.25	0.18	0.11	0.0881	0.0541
b_1	0.88	-0.99	-1.19	-1.35	-2.88	-3.80
b_2	-0.15	0.82	0.85	0.79	1.26	1.42
b_3	0.15	0.017	0.0045	0.0022	-0.0421	-0.0551

a $\sigma_y = (a_1 \ln x + a_2)x$

$$\sigma_z = \frac{1}{2.15} \exp(b_1 + b_2 \ln x + b_3 \ln^2 x)$$

x = downwind distance, m.

Table 2.2. Coefficients for specifying two systems of dispersion parameters for six stability categories[9]

System	Coefficient[a]	Atmospheric stability category					
		A	B	C	D	E	F
Julich[9]	p_y	0.66	0.66	0.63	0.53	0.41	7.56
	q_y	0.83	0.83	0.80	0.80	0.87	0.52
(H ≤ 75m)	p_z	0.14	0.14	0.21	0.26	0.13	0.56
	q_z	1.09	1.09	0.98	0.89	0.83	0.55
Julich[9]	p_y	0.34	0.37	0.40	0.43	0.46	7.56
	q_y	1.00	0.94	0.88	0.82	0.76	0.52
	p_z	0.037	0.076	0.16	0.32	0.66	1.37
(H > 75m)	q_z	1.28	1.12	0.96	0.88	0.63	0.47
			(B$_2$)	(B$_1$)	(C)		(D)
Brookhaven[11]	p_y		0.40	0.36	0.32		0.31
(measured at H = 108m)	q_y		0.91	0.86	0.78		0.71
	p_z		0.41	0.33	0.22		0.062
	q_z		0.91	0.86	0.78		0.71

$^a \sigma_y = p_y x^{q_y}$

$\sigma_z = p_z x^{q_z}$

x = downwind distance, m.

h_t = maximum terrain height (above the stack base) between the release point and the point for which the calculation is being made ($h_t \geq 0$),

C = downwash correction factor.

Values of h_{pr} are usually calculated through the use of models such as those suggested by Briggs.[16] Models have also been proposed for calculating C.[15] The need to include estimates of h_{pr}, h_t, and C in estimates of H for a given radiological assessment will depend on the particular facility under consideration and its location.

2.1.1.3 _Radioactive decay_. The Gaussian model, as expressed by Eqs. (2.1) and (2.2), assumes no change in the species of material released as it is being transported downwind. This, of course, is not the case when the released material is radioactive. As a result, air concentrations of radionuclides calculated using the model should be corrected to account for decay of the material during transit and subsequent buildup of daughter products.[17]

2.1.2 _Other atmospheric dispersion models_

There are a number of situations commonly encountered in radiological assessments for which the Gaussian plume model may not apply or may be difficult to parameterize. These include instances involving complex terrain, long range transport, time-varying meteorology, and variable release rates. As a result, a large number of more complex, seemingly more realistic, dispersion models have been developed. However, to run properly, these models often require a much more extensive input data base than the Gaussian model, a computer with large storage capacity, and a long computer running time for each simulation desired. These

conditions can severely limit the practicality of using a more complex model in assessment activites. The trajectory model[18] is gaining acceptance as a tool for estimating dispersion on a continental scale.[19] In general, however, more field data are needed to specify clearly when a given complex model should supplement the Gaussian plume model in radiological assessments.

2.2 Deposition

Particulates and reactive gases may deposit on the surface of the earth through the processes of dry and wet deposition. These processes affect doses to man from atmospheric releases of radionuclides in two ways:

1. deposited material serves as a source of surface and/or food chain contamination;

2. deposition results in a reduction in the amount of material transported downwind in air through plume depletion.

2.2.1 Dry deposition

Dry deposition is the process by which particles and reactive gases deposit on various surfaces (soil, grass, leaves, etc.) via impingement, electrostatic interactions, chemical reactions, and other processes. The rate of deposition $d(Ci/m^2 \cdot sec)$ is given by[20]

$$d = \chi v_d \quad , \tag{2.4}$$

where

χ = ground-level air concentration (Ci/m^3),

v_d = deposition velocity (m/sec).

The deposition velocity defined in Eq. (2.4) is a transfer factor relating an air concentration to a surface deposition rate. Field measurements of v_d, however, are generally based on measured concentrations in vegetation cut at a specific height above ground.[21,22] Thus, an estimate of v_d appropriate for the total deposit on a unit area basis is derived from a v_d specific for deposition onto vegetation by[17]

$$v_d \text{ (total)} = v_d \text{ (vegetation)}/R \qquad (2.5)$$

where

    R = fraction of the total material being deposited which is intercepted by the surface of interest.

Values of v_d (vegetation) for forage grasses have been determined:[23]

1. 0.02 m/sec for reactive gases (molecular iodine),

2. 0.001 m/sec for small particulates (<4 μm diam), and

3. 0.0001 m/sec for relatively unreactive gases (CH_3I).

Using a mean forage grass interception fraction value of R = 0.57 (Sect. 3) results in the following values of v_d (total):

1. 0.035 m/sec for reactive gases,

2. 0.0018 m/sec for small particulates, and

3. 0.00018 m/sec for relatively unreactive gases.

If the plume traverses surfaces other than grasslands, other values of v_d (total) should be considered if they are available.[22]

2.2.2 Wet deposition

Wet deposition is the process by which particles or gases are scavenged from a plume by rain or snow and deposited on ground surfaces. The rate of deposition on ground surfaces from these processes, ω, is often estimated by[24]

$$\omega = \Phi \chi_{va} L \quad , \tag{2.6}$$

where

Φ = fraction of material scavenged from a vertical column of air per unit time (sec^{-1}),

χ_{va} = average concentration in vertical column (Ci/m^3),

L = height of vertical column (m).

The value chosen for L is often the depth of the tropospheric mixing layer.[24]

It has also been suggested that the wet deposition rate for long-term average situations can be calculated by using Eq. (2.4) and a wet deposition velocity v_w which is given by[11]

$$v_w = \frac{k_o p}{\chi_o} = w_r p \quad , \tag{2.7}$$

where

k_o = surface level concentration in the precipitation (Ci/m^3),

χ_o = surface level air concentration (Ci/m^3),

p = amount of precipitation per unit time (m/year),

$w_r = \dfrac{k_o}{\chi_o}$ = washout ratio.

A comparison of results from these two methods (Eqs. 2.6 and 2.7) using annual average meteorological data for Oak Ridge, Tennessee, indicates that beyond 1 km downwind from the source there is little difference between them.[25] For considering wet deposition from a single event, however, Eq. (2.6) is the more appropriate method to use.[11]

2.2.3 Plume depletion

Deposition processes deplete the airborne plume as it travels downwind. The most common method of accounting for this removal is

reducing the release rate of the material. In the case of the Gaussian model [Eq. (2.1)] this new release rate Q' is given for dry deposition by

$$Q' = Q \exp\left\{-\left(\frac{2}{\pi}\right)^{1/2} \frac{v_d(\text{total})}{u} \int_0^x \frac{\exp\left[-\frac{1}{2}\left(\frac{H}{\sigma_z}\right)^2\right]}{\sigma_z} dx'\right\} . \quad (2.8)$$

A number of other methods of accounting for plume depletion have been proposed, but until validation data become available for determining which one is really best, the continued use of Eq. (2.7) seems reasonable.[25]

If wet deposition is estimated using Eq. (2.6) and a scavenging coefficient Φ', the reduced release rate is given by[24]

$$Q' = Q \exp(-\Phi't) \quad , \quad (2.9)$$

where t is the time required for the plume to reach a given point downwind, and Φ' is averaged over the entire time period of interest, including periods without precipitation. If Eq. (2.7) is used to estimate wet deposition, however, Eq. (2.8) may be used to estimate plume depletion by substituting v_w for v_d.

2.2.4 Gravitational settling

Equations (2.4) and (2.8) apply when the gases or particulates in the plume are small enough not to be significantly affected by gravity as they travel downwind. If the plume contains larger particles, however, the gravitational settling may be approximated by tilting the plume downward. This is done by replacing the effective stack height H by[24]

$$H - \frac{v_g x}{u} \quad ,$$

where

v_g = gravitational fall velocity (m/sec),

x = downwind distance of interest (m).

It must be specified that $H - \frac{v_g x}{u} \geq 0$.

2.3 Duration of Release

The point-source Gaussian plume model, Eq. (2.1) and (2.2) with the modifications discussed above, attempts to define a mean concentration discussed above, attempts to define a mean concentration field relative to a fixed location over some relatively long period of time.[5] The averaging time of the model is that of the diffusion data upon which dispersion parameters are based. Not all sets of dispersion parameters have the same release or sampling time. For example, the Pasquill-Gifford curves are based on a sampling duration of 3 minutes[10] while the Julich parameters are based on releases of generally 1 hr duration.[9]

Often times one is interested in calculating concentrations for averaging times other than the one applicable for the dispersion parameters being used. It has been suggested[5,10] that this can be accomplished by modifying σ_y according to the following relationship:

$$\frac{\sigma_{yA}}{\sigma_{yB}} = \left(\frac{t_A}{t_B}\right)^q \qquad (2.10)$$

where

σ_{yA} = horizontal dispersion parameter average over some time period of interest, t_A, and

σ_{yB} = original value of the horizontal dispersion parameter based on diffusion data averaged over some time period t_B.

Reasonable values for q have been found to be in the range of 0.25 - 0.3 for 1 hr < t_A < 100 hr and approximately equals 0.2 for 3 min < t_A < 1 hr.[5,10] A similar relationship could be applied to σ_z, but such variations in σ_z should not extend beyond a few kilometers downwind of the release point.

For calculating average concentrations over long time periods, such as a season or a year, Eq. (2.2) is generally used with a weighting for the fraction of the time that the wind blows toward the point of interest.[5] Joint frequency distributions of wind direction, wind speed, and atmospheric stability can be constructed for many locations from climatological records.

The methods discussed in this chaper must be further modified to calculate dispersion from sudden, explosive, or very short term (i.e., t_B < 3 min) releases of material to the atmosphere. In general, the values of σ_y and σ_z used for such releases are considered to be different than the values presented above for longer time releases.[1] Also, dispersion in the downwind direction must be taken into consideration for very short term releases. However, virtually nothing is known about the downwind dispersion parameter, σ_x.[5] Accidental releases from breeder reactor facilities have generally been considered to be of long enough duration for the methods considered in this section to be applied.[26]

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3. TERRESTRIAL FOOD CHAIN TRANSPORT

Laura M. McDowell-Boyer
C. F. Baes III

A number of computer codes are available with which terrestrial transport of radionuclides through food chains following release to the atmosphere may be quantitatively estimated. The majority of these models were derived from the HERMES computer code,[1] which was developed for assessing transport in chronic, or routine, release situations.[2] Distinct from these HERMES-based models is TERMOD,[3] a model developed for assessing terrestrial transport under routine release conditions or following acute, or accidental, radionuclide releases to the environment. All of these models, however, utilize equilibrium transfer coefficients to quantify the transport between food chain compartments following deposition of airborne radionuclides on soil or interception by vegetation. The purposes of this section are first to define the parameters for which values are needed in currently implemented terrestrial transport models at Oak Ridge National Laboratory (ORNL), and secondly, to present the parameter values used in these models.

3.1 Terrestrial Models

Terrestrial transport models currently in use at ORNL for assessing routine radionuclide releases are implementations of models provided in the October 1977 draft of the U. S. Nuclear Regulatory Commission (NRC) Regulatory Guide 1.109, Appendix C.[4] These latter models were originally provided for assessing transport of light-water reactor effluents but have been used to assess terrestrial nuclide transport in

general. For breeder reactor effluents, it may be desirable to consider resuspension of nuclides following discontinuation or interruption of plant operation, when this phenomenon may be a significant source of airborne nuclides, and to consider buildup of daughter nuclides as was mentioned previously (Sect. 2.1.1.3). These models consider the concentration of radionuclides in vegetation as a result of deposition onto plant tissues and root uptake of activity initially deposited on soil, as well as concentrations in milk and beef following grazing on contaminated vegetation by dairy and beef cattle. Many shortcomings of equilibrium models such as the following are recognized, and thus, work is underway to clarify, evaluate, and possibly rectify some of these shortcomings. However, at present, our discussions will be restricted to these currently implemented models.

3.1.1 Concentrations in vegetation

The following equation[4] is used for estimating the concentration $C_i^v(r, \theta)$ of nuclide i in and on vegetation at the location (r, θ)

$$C_i^v(r, \theta) = d_i(r, \theta) \frac{R[1 - \exp(-\lambda_{Ei}t_e)]}{Y_v \lambda_{Ei}} + \frac{B_{iv}[1 - \exp(-\lambda_i t_b)]}{P\lambda_i} \exp(-\lambda_i t_h), \quad (3.1)$$

where

$C_i^v(r, \theta)$ is measured in pCi/kg;

$d_i(r, \theta)$ is the deposition rate of radionuclide i onto ground at location (r, θ), in pCi/m^2-hr;

R is the fraction of depositing activity intercepted by crops, dimensionless;

λ_i is the radioactive decay constant of nuclide i, in hr^{-1};

λ_{Ei} is the effective removal rate constant for radionuclide i from crops, in hr^{-1}, where $\lambda_{Ei} = \lambda_i + \lambda_w$,

λ_w is the removal rate constant for physical loss by weathering, in hr^{-1};

t_e is the time period that crops are exposed to contamination during the growing season, in hours;

Y_v is the agricultural productivity (yield) of the edible portion, in kg/m^2;

B_{iv} is the concentration factor for uptake of radionuclide i from soil by edible parts of crops, in pCi/kg plant tissue per pCi/kg dry soil;

t_b is the period of long-term buildup for activity in soil, in hours;

P is the effective "surface density" for the top 15 cm of soil, in kg (dry soil)/m^2;

t_h is a holdup time that represents the time interval between harvest and consumption of the food, in hours.

Regulatory Guide 1.109 (ref. 4) makes a distinction between the value of t_e, Y_v, and t_h appropriate for forage grasses and those appropriate for crops and leafy vegetables. The ORNL implementation of Eq. (3.1) (ref. 5) further distinguishes the C_i^v calculated for pasture grasses from a C_i^v calculated for fresh vegetables and produce consumed by man by inputting separate values of B_{iv} and R or R/y_v for each of these categories. For pasture grasses, $Y_{v(1)}$ and $B_{iv(1)}$, based on dry weight concentrations in both vegetation and soil, are used for Y_v and B_{iv}. When calculating concentrations in fresh produce consumed by man, $Y_{v(2)}$ and $B_{iv(2)}$ are used

for Y_v and B_{iv} values, based on fresh-weight concentrations in vegetation and dry-weight soil concentrations of a nuclide.

3.1.2 Concentrations in milk

The concentration of radionuclide i in milk depends upon the amount and contamination level of the feed consumed by the animal. The concentration of radionuclide i in the animal's feed is calculated by use of the equation

$$C_i^V(r, \theta) = f_p f_s C_i^P(r, \theta) + (1 - f_p f_s)\, C_i^S(r, \theta) \qquad (3.2)$$

where

$C_i^V(r, \theta)$ is the concentration of radionuclide i in the animal's feed, in pCi/kg;

$C_i^P(r, \theta)$ is the concentration of radionuclide i on pasture grass (calculated using Eq. (3.1) with $t_h = 0$), in pCi/kg;

$C_i^S(r, \theta)$ is the concentration of radionuclide i in stored feeds (calculated using Eq. (3.1) with $t_h = 2160$ hours (90 days), in pCi/kg;

f_p is the fraction of the year that animals graze on pasture, dimensionless;

f_s is the fraction of daily feed that is pasture grass when the animals graze on pasture, dimensionless.

Using the value of $C_i^V(r, \theta)$ calculated by use of this equation, the concentration of radionuclide i in milk is estimated as

$$C_i^M(r, \theta) = F_m C_i^V(r, \theta) Q_F \exp(-\lambda_i t_f) , \qquad (3.3)$$

where

$C_i^M(r, \theta)$ is the concentration in milk of nuclide i in pCi/liter;

$C_i^V(r, \theta)$ is the concentration of radionuclide i in the animal's feed, in pCi/kg;

F_m is the average fraction of the animal's daily intake of radionuclide i which appears in each liter of milk, in days/liter;

Q_F is the amount of dry feed consumed by the animal per day, in kg/day;

t_f is the average transport time of the activity from the feed into the milk and to the receptor, in days;

λ_i is the radiological decay constant of nuclide i, in days^{-1}.

3.1.3 Concentrations in meat

The radionuclide concentration in meat (usually beef) depends upon the amount and contamination level of the feed consumed by the animal, as in the milk pathway. Using the value of $C_i^V(r, \theta)$ as calculated in Eq. (3.2), the radionuclide concentration in meat is estimated as

$$C_i^F(r, \theta) = F_f C_i^V(r,, \theta) Q_F \exp(-\lambda_i t_s) , \tag{3.4}$$

where

$C_i^F(r, \theta)$ is the concentration of nuclide i in animal flesh, in pCi/kg;

F_f is the fraction of the animal's daily intake of nuclide i which appears in each kilogram of flesh, in days/kg;

$C_i^V(r, \theta)$ is the concentration of radionuclide i in the animal's feed, in pCi/kg;

Q_F is the amount of dry feed consumed by the animal per day, in kg/day;

λ_i is the radiological decay constant of nuclide i, in days^{-1};

t_s is the average time (days) from slaughter to consumption.

For concentrations in beef, it is assumed that beef cattle are on open pasture for the same grazing periods as given for milk cattle.

3.2 Terrestrial Transport Input Parameters

Following are lists of input parameter values currently used at ORNL in assessing terrestrial transport resulting from routine releases. These values were derived from empirical data obtained from an ongoing review of available literature,[5] and represent average values over a wide range of experimental and environmental conditions. Values listed for the transfer coefficients, $B_{iv(1)}$, $B_{iv(2)}$, F_m, and F_f, reflect the current status of this literature review of element-specific data pertinent to terrestrial transport, and may change when new references become available and are reviewed.

Not all transfer coefficients reported here have been evaluated in this project. Values given for these parameters are taken from other widely used documents.[6,7] The derivations of transfer coefficients for nuclides which have been evaluated are element, rather than nuclide, specific, so that the data base for some nuclides is not severely restricted.

The parameters used in the terrestrial transport models described are designed to represent annually averaged values at equilibrium over a wide range of environmental conditions. Thus, these average parameters are best suited for generic assessments, but may be used to preliminarily assess specific sites when site specific parameters are not available. Empirical data used in deriving values for these parameters may not

always reflect annual average or equilibrium conditions, nor field
conditions when laboratory or otherwise controlled experiments were
conducted. For technetium, appreciable uptake of the element from soil
by plant roots may result in nonequilibrium soil concentrations in
laboratory studies concerning technetium uptake. For this reason and
because the transport model used does not consider soil loss of nuclides
via crop harvest or leaching, the assessment of technetium transport
using the concentration factor approach may result in overestimates of
plant concentrations. Furthermore, empirical data may not adequately
reflect the true distribution of values associated with each parameter
under various conditions. Thus, caution should be used in the interpretation
of parameter values presented and results generated in implementation of
the models.

Statistical distributions have been estimated for Y_v, f_p, f_s, Q_F,
and P,[8] and values describing the distributions for each of these
parameters are provided here. For B_{iv}, F_m, and F_f, an arithmetic mean
value was calculated from data obtained from each reference cited.
These mean values were used in deriving an arithmetic mean for all
references collected for a particular element. Statistical distri-
butions have not yet been developed for these latter parameters.

Terrestrial transport of ^3H and ^{14}C is addressed in Sect. 8 of this
report, and thus, transfer coefficients for these nuclides are not
included in this section.

3.2.1 Agricultural productivity by unit area, $Y_{v(1)}$ and $Y_{v(2)}$

Agricultural productivity, in kg/m^2, is given in dry weight in
this report for pasture grasses, $Y_{v(1)}$, and in fresh weight for leafy

vegetables and produce ingested directly by man, $Y_{v(2)}$. This method of reporting parameter values for Y_v has been found to be most directly useful in the transport models used. Table 3.1 presents mean values of $Y_{v(1)}$ and $Y_{v(2)}$ derived from the review and analysis of agricultural productivity by Hoffman and Baes along with the μ and σ, representing the mean and standard deviation of the log-transformed distribution, respectively.

3.2.2 Interception fractions for above-ground plant portions, R_1 and R_2

The fraction of atmospherically depositing radionuclides intercepted and initially retained on above-ground portions of either forage crops or leafy vegetables and fresh produce ingested by man is symbolized by R_1 and R_2, respectively, where R_1, $R_2 \leq 1$. The remaining fraction deposited on the soil and surfaces other than the vegetation is merely $(1-R_1)$ and $(1-R_2)$.

Measurements of R_1 for specific values of $Y_{v(1)}$ have been made and reviewed by Hoffman and Baes and the resultant mean value of $R_1/Y_{v(1)}$ is given in Table 3.1. Measurements of the interception fraction R_2 specific for edible portions of leafy vegetables and fresh produce ingested directly by man are unavailable. It is expected that the value of R_2 will be less than that of R_1 since vegetable crops are usually cultivated in rows. On a unit area basis, this spacing of vegetable crops exposes more surface soil than the dense spacing of forage crops. Therefore, a value of 0.2 for R_2, provided in the USNRC Regulatory Guide 1.109 (ref. 4) is currently being used. These R_1 and R_2 values are being used for both wet and dry deposition.

Table 3.1. Estimated values of above-ground agricultural productivity and interception fractions for forage grasses and edible portions of vegetable crops

Parameter	Mean value (kg/m^2)	μ	σ	Ref.
R_1/Y_{v1} (Y_{v1} measured in dry weight- forage grasses)[a]	2.0	0.61	0.44	8
Y_{v2} measured in fresh weight (leafy vegetables)[b,c]	1.9	-0.73	0.46	8
Y_{v2} measured in fresh weight (nonleafy vegetables)[b,d]	0.57	0.43	0.52	8
R_2 (vegetables)[e]	0.20			4

[a]R_1 defined as interception fraction for forage grasses.

[b]The estimates of the mean are the inverse of values derived by Hoffman and Baes[8] (μ and σ are specific for $1/Y_{v2}$).

[c]The distribution described by Hoffman and Baes[6] is based on edible portions of cabbage lettuce, and spinach.

[d]The distribution described by Hoffman and Baes[6] is based on edible portions of broccoli, cauliflower, green peas, lima beans, and sweet corn.

[e]R_2 defined as interception fraction for edible portions of leafy vegetables and fresh produce. No statistical parameters are available.

3.2.3 Plant/soil bioaccumulation factor, B_{iv}

Tables 3.2 and 3.3 contain values of $B_{iv(1)}$ and $B_{iv(2)}$, representing the transfer of elements from soil to grass leaves and portions of vegetables and produce considered edible by man, respectively. The following definitions of $B_{iv(1)}$ and $B_{iv(2)}$ apply:

$$B_{iv(1)} = \frac{\text{radionuclide concentration in entire above-ground portion of plant at maturity per unit dry wt}}{\text{radionuclide concentration in soil per unit dry wt.}}$$

$$B_{iv(2)} = \frac{\text{radionuclide concentration in edible portion of plant at maturity per unit fresh wt.}}{\text{radionuclide concentration in soil per unit dry wt.}}$$

As was mentioned previously, arithmetic means of results obtained from empirical studies concerning root uptake for each element are currently being used for B_{iv} values.

The parameter B_{iv} is interpreted as the elemental concentration in plant tissues at maturity resulting from an equilibrium soil concentration to which the root are exposed during the growing season. For Tc, however, appreciable uptake of the element from soil by plant roots may result in non-equilibrium soil concentrations in laboratory studies concerning Tc uptake.[14] Because no field data have been reported for Tc, the appropriate B_{iv} cannot be directly derived. The $B_{iv(2)}$ for Tc (Table 3.2) is simply a conversion of the $B_{iv(2)}$ value to represent a dry-weight concentration in forage. No distinction between plant portions or species was made in this case, due to the lack of empirical data on different plant types.

Table 3.2. Values of $B_{iv(1)}$ [(element)$_{forage}$:
(element)$_{soil}$] for forage and feed

Element	$B_{iv(1)}$[a]	References
Na	2.1×10^{-1}	6[b]
P	4.4	6[b]
S	2.4	6[b]
Mn	1.2×10^{-1}	6[b]
Fe	2.6×10^{-3}	6[b]
Co	3.8×10^{-2}	6[b]
Ni	7.6×10^{-2}	6[b]
Zn	1.6	6[b]
Sr	1.2	9-12
Y	1.1×10^{-2}	9,13
Zr	6.8×10^{-4}	6[b]
Nb	3.8×10^{-2}	6[b]
Tc	2.0×10^{2}	14
Ru	1.7×10^{-1}	9,13
Ag	6.0×10^{-1}	6[b]
Sb	4.4×10^{-2}	6[b]
Te	5.2	6[b]
I	2.0×10^{-1}	15
Cs	1.5×10^{-1}	9,11,13,16-20
Ce	3.9×10^{-2}	9,13
Pm	1.3×10^{-3}	13
Sm	1.0×10^{-2}	6[b]
Eu	1.0×10^{-2}	6[b]
Pb	1.4×10^{-1}	21-24
Bi	6.0×10^{-1}	6[b]
Po	4.2×10^{-3}	25

Table 3.2. (continued)

Element	$B_{iv(1)}{}^a$	References
Ra	9.1×10^{-2}	26-28
Ac	1.0×10^{-2}	6[b]
Th	2.7×10^{-3}	29
Pa	1.0×10^{-2}	6[b]
U	6.1×10^{-3}	29,30
Np	1.0×10^{-2}	6[b]
Pu	2.0×10^{-3}	11,13,30-34
Am	2.1×10^{-3}	29-31,34
Cm	4.8×10^{-4}	32

[a]$B_{iv(1)}$ is derived for dry-weight concentrations in forage, hay, or feed and in soil.

[b]Derived by converting fresh-weight plant concentrations (C_p) to dry-weight, assuming 25% dry matter content, and dividing this value by dry-weight soil concentrations, C_s.

[c]Converted from a given fresh-weight concentration factor to dry-weight, by assuming 25% dry matter for plants.

[d]See preceding discussion.

Table 3.3. Values of $B_{iv(2)}$ [(element) edible tissue : (element)$_{soil}$] for vegetables, fruits, and grains

Element	$B_{iv(2)}$[a]	References
Na	5.2×10^{-2}	6[b]
P	1.1	6[b]
S	5.9×10^{-1}	6[b]
Mn	2.9×10^{-2}	6[b]
Fe	6.6×10^{-4}	6[b]
Co	9.4×10^{-3}	6[b]
Ni	1.9×10^{-2}	6[b]
Zn	4.0×10^{-1}	6[b]
Sr	2.9×10^{-1}	9,11,12,35
Y	4.3×10^{-3}	9
Zr	1.7×10^{-4}	6[b]
Nb	9.4×10^{-3}	6[b]
Tc	5.0×10^{1}	14[c]
Ru	1.6×10^{-2}	9
Ag	1.5×10^{-1}	6[b]
Sb	1.1×10^{-2}	6[b]
Te	1.3	6[b]
I	5.5×10^{-2}	15
Cs	1.1×10^{-2}	9,11,19,20,35,36
Ce	6.2×10^{-3}	9
Pm	2.5×10^{-3}	6[b]
Sm	2.5×10^{-3}	6[b]
Eu	2.5×10^{-3}	6[b]
Pb	3.9×10^{-3}	21,23,37,38
Bi	1.5×10^{-1}	6[b]
Po	2.6×10^{-4}	25

Table 3.3. (continued)

Element	$B_{iv(2)}{}^{a}$	References
Ra	1.3×10^{-2}	26,27,39,40
Ac	2.5×10^{-3}	6[b]
Th	3.5×10^{-4}	41
Pa	2.5×10^{-3}	6[b]
U	2.9×10^{-4}	30,41
Np	2.5×10^{-3}	6[b]
Pu	2.2×10^{-4}	11,30,33,42-44
Am	4.0×10^{-4}	29
Cm	1.7×10^{-3}	15

[a]$B_{iv(2)}$ is derived for a fresh-weight concentration in edible plant tissues and a dry-weight concentration in soils.

[b]Derived by dividing the fresh-weight plant concentration (C_p) by dry-weight soil concentrations (C_s).

[c]See preceding discussion.

3.2.4 Milk transfer coefficient, F_m

The milk transfer coefficient, F_m, represents the fraction of the total daily intake of a nuclide which is transferred to a liter of the cow's milk at equilibrium. Values of F_m can be derived from a variety of reported data according to the methodologies described by Ng et al.[7]

Determinations of F_m (Table 3.4) were made from literature sources reporting empirically-derived results, whenever available. However, for many elements, the well-documented literature review by Ng et al.[7] was the primary literature source. References in which the chemical form of the nuclide administered orally to the cow was clearly atypical of forms found in the environment were excluded from the analysis of F_m. As for the analysis of B_{iv}, the values of F_m given in Table 3.4 are element- rather than nuclide-specific.

3.2.5 Meat transfer coefficient, F_f

The meat-transfer coefficient F_f represents the fraction of the total daily intake of a nuclide which is transferred to a kilogram of muscle in the meat producing animal at equilibrium. It is assumed that equilibrium conditions exist when slaughter occurs.

A review of the available literature allowed the derivation of values of F_f for beef cattle (Table 3.5). Values of F_f based on immature cattle (less than 6 months of age) were exluded from analysis when data for adult cattle were available. Data for other ruminant species were included when literature references for cattle were unavailable.

Table 3.4. Values of F_m [(element)$_{milk}$: elemental daily intake], day/liter

Element	$F_m{}^{\alpha}$	References
Na	3.5×10^{-2}	7
P	1.6×10^{-2}	7
S	1.6×10^{-2}	7
Mn	8.4×10^{-5}	7
Fe	5.9×10^{-5}	7
Co	2.0×10^{-3}	7
Ni	1.0×10^{-2}	7
Zn	1.0×10^{-2}	7
Sr	2.4×10^{-3}	45-48
Y	2.0×10^{-5}	7
Zr	8.0×10^{-2}	7
Nb	2.0×10^{-2}	7
Tc	9.9×10^{-3}	7
Ru	6.1×10^{-7}	7
Ag	3.0×10^{-2}	7
Sb	2.0×10^{-5}	7
Te	2.0×10^{-4}	7
I	1.0×10^{-2}	7,8
Cs	5.6×10^{-3}	49-51
Ce	2.0×10^{-5}	7
Pm	2.0×10^{-5}	7
Sm	2.0×10^{-5}	7
Eu	2.0×10^{-5}	7
Pb	9.9×10^{-5}	52-59
Bi	5.0×10^{-4}	7
Po	1.2×10^{-4}	58
Ra	5.9×10^{-4}	59
Ac	2.0×10^{-5}	7
Th	5.0×10^{-6}	7

Table 3.4. (continued)

Element	$F_m{}^a$	References
Pa	5.0×10^{-6}	7
U	1.2×10^{-4}	60
Np	5.0×10^{-6}	7
Pu	4.5×10^{-8}	61,62
Am	2.0×10^{-5}	7
Cm	2.0×10^{-5}	7

[a]The F_m is derived from fresh-weight concentrations in milk and dry-weight concentrations in forage, feed, or hay.

Table 3.5.   Values of F_f [(element)$_{beef}$:   elemental
daily intake], day/kg

Element	$F_f{}^a$	References
Na	3.8×10^{-2}	6[b]
P	5.7×10^{-2}	6[b]
S	1.3×10^{-1}	6[b]
Mn	1.0×10^{-3}	6[b]
Fe	5.0×10^{-2}	6[b]
Co	1.7×10^{-2}	6[b]
Ni	6.7×10^{-3}	6[b]
Zn	3.8×10^{-2}	6[b]
Sr	3.0×10^{-4}	15
Y	5.8×10^{-3}	6[b]
Zr	4.3×10^{-2}	6[b]
Nb	3.5×10^{-1}	6[b]
Tc	8.7×10^{-3}	6[b]
Ru	1.8×10^{-3}	15
Ag	2.2×10^{-2}	6[b]
Sb	5.0×10^{-3}	6[b]
Te	9.6×10^{-2}	6[b]
I	7.0×10^{-3}	15
Cs	1.4×10^{-2}	63
Ce	6.0×10^{-4}	15
Sm	6.3×10^{-3}	6[b]
Eu	6.0×10^{-3}	6[b]
Pb	9.1×10^{-4}	57
Bi	1.7×10^{-2}	6[b]
Po	4.0×10^{-3c}	64-66

Table 3.5. (continued)

Element	$F_f{}^a$	References
Ra	5.5×10^{-4}	65,67[c]
Ac	1.6×10^{-6}	d
Th	1.6×10^{-6}	d
Pa	1.6×10^{-6}	d
U	1.6×10^{-6}	d
Np	1.6×10^{-6}	d
Pu	4.1×10^{-7}	15,62
Am	1.6×10^{-6}	15
Cm	1.6×10^{-6}	15

[a]The F_f is derived from fresh-weight concentrations in beef and dry-weight concentrations in forage, feed, or hay.

[b]Derived by converting fresh-weight plant concentrations (C_p) to dry-weight, assuming 25% dry matter content, such that

$$F_f = \frac{C_{meat} \text{ (fresh)}}{C_p \text{ (dry)}} \times \frac{1}{Q_F} \; .$$

[c]Values given are based on sheep, caribou, and reindeer data and the associated intake rates for each.

[d]Values are assumed to be the same as those for curium and americium.

3.2.6 Animal feed consumption (Q_F), grazing patterns (f_p, f_s), and effective soil surface density (P)

Values used for the parameters Q_F, f_p, f_s, and P (defined in Section 3.1) are the means of their respective statistical distributions as described by Hoffman and Baes. These means are given in Table 3.6, along with the associated μ and σ for the log-transformed distribution.

3.2.7 Time parameters t_e, t_b, t_h, t_f, and t_s

The parameters t_e, t_h, t_f, and t_s have all been defined in Section 3.1, and the values used at ORNL are listed in Table 3.7. The values listed were taken from the draft Regulatory Guide.[4] The parameter t_b, representing the period of long-term buildup for activity in soil, was not assigned a specific value. The value of t_b is left to the user's discretion.

Table 3.6. Values of Q_F, f_p, f_s, and P used in
terrestrial transport models at ORNL

Parameter	Mean	μ	σ	S.D.	Units	Reference
$Q_F{}^a$	15.6			2.6	kg/day (dry)	8
$f_p{}^a$	0.40			0.22		8
$f_s{}^a$	0.43			0.13		8
P^b	215	5.36	0.11		kg/m^2	8

[a]Parameter normally distributed; therefore, standard deviation (S.D.) is given.

[b]Distribution for P derived from a lognormal distribution of soil bulk density $\rho(g/cm^3)$.

Table 3.7. Values of t_e, t_h, t_f, and t_s used
in terrestrial transport models at ORNL

Parameter	Value	Units	Reference
$t_e{}^a$: $t_{e(1)}$	720	hr	4
$t_{e(2)}$	1440	hr	
$t_h{}^b$: $t_{h(1)}$	0	hr	4
$t_{h(2)}$	2160	hr	4
$t_{h(3)}$	336	hr	4
$t_{h(4)}$	336	hr	4
t_f	4	day	4
t_s	20	day	4

$^a t_{e(1)}$ applies to forage grasses; $t_{e(2)}$ applies to crops and leafy vegetables.

$^b t_{h(1)}$ applies to forage; $t_{h(2)}$ applies to stored feed for animals; $t_{h(3)}$ applies to leafy vegetables; and $t_{h(4)}$ applies to produce.

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4. SURFACE WATER TRANSPORT

C. A. Little

This section will briefly discuss models which we feel are best suited to assess radionuclide releases from breeder reactors to hydrologic (surface water) systems. The models which we consider are two-dimensional, either longitudinal-transverse or longitudinal-vertical. The models also differ in their capability to account for sorptive effects on the pollutant. The Yotsukura-Sayre model (longitudinal-transverse) does not consider sorptions, but the SERATRA (longitudinal-vertical) and FETRA (longitudinal-transverse) models do consider sorption and sediment transport. These models and their parameter needs will be discussed separately.

4.1 Model Without Sorption

The Yotsukura-Sayre model,[1] a continuous release equilibrium model, is basically the same as that of Yotsukura and Cobb.[2] These models employ an orthogonal curvilinear coordinate system to describe the geometrical configuration of the channel. In its most fundamental form the model can be expressed as follows:[3]

$$uD \frac{\partial C_i}{\partial x} = \frac{\partial}{\partial y} \left(K_y D \frac{\partial C_i}{\partial y} \right) - (\lambda D) C_i , \qquad (4.1)$$

where

C_i = concentration of radionuclide $i (Ci\ m^{-3})$,

u = stream velocity $(m\ sec^{-1})$,

D = stream depth (m),

x = distance downstream (m) ,

y = distance across stream (m),

K_y = lateral turbulent diffusion coefficient (m^2sec^{-1}), and

λ = radioactive decay constant = 0.693/half life (sec^{-1}).

Of these parameters, all but K_y can be measured directly in the stream of concern. However, Yotsukura and Cobb[2] showed that the variable diffusion factor, $K_y u D^2$, may be replaced by a constant factor, $\overline{K_y u D^2}$ where

$$\overline{K_y u D^2} = \frac{1}{Q} \int_0^Q K_y u D^2 dq \quad , \tag{4.2}$$

Q is the total river discharge and q is the transverse cumulative discharge measured outward from the near shore. Further, Yotsukura and Cobb[2] suggest that K_y can be determined using the formula of Elder[4]:

$$K_y = \beta D u^* \quad , \tag{4.3}$$

where β is a unitless constant, D is stream depth in m and u^* is the average shear velocity in the y dimension. Therefore, the only constant which may be considered not site specific for the model is the constant β. The value of β in flume studies has been shown to range from about 0.11 to 0.23.[2] A value of β = 0.72 was found in the Columbia River,[5] but Fischer[6] showed that secondary currents could increase the value of β by as much as a factor of 10. A value of β = 0.6 has been observed in the Missouri River.[7]

A model user who is interested in predicting the maximum concentration downstream from the same point release should choose a value of β = 0. A zero β would result in no diffusion and, as seen from Eq. (4.1), would simplify the prediction of activity concentration to a function of decay and downstream distance. This would effectively eliminate the consideration of transverse (or lateral) distance and maximize the activity concentration at any point in midstream.

A model user who is not interested in a conservative estimate will be faced with deciding which value of β to use. From the review of Yotsukura and Cobb[2] we conclude that β may range from approximately 0.1 to 0.72 (and perhaps higher). We would suggest, then, that a user interested in accurate predictions of pollutant concentrations in water should use a value of β greater than 0.1 and less than 1.0.

4.2 Models with Sorption

If the pollutant being assessed is not in a soluble form or has a marked tendency to adsorb to sediments, as do many actinides, the user may wish to use a model which is capable of predicting sediment or particulate contaminant transport. For such an application, we suggest considering usage of the FETRA model discussed by Onishi and Wise[8] in spite of the large number of input parameters it requires. FETRA and a similar model, SERATRA,[9] both estimate sediment and pollutant concentrations downstream from some release site as a function of time after the release. However, SERATRA estimates concentrations longitudinally and vertically, while FETRA considers longitudinal and lateral (transverse) distribution of the pollutant. These models are both much more complex than the Yotsukura model mentioned above, and therefore, SERATRA and FETRA will be discussed in general terms, neglecting specific parameters.

Both SERATRA and FETRA are time-dependent models which consist of three submodels: (1) a sediment transport model, (2) a dissolved contaminant transport model, and (3) a particulate contaminant transport model.

The sediment transport sections of FETRA and SERATRA are capable of accounting for effects of: (1) convection and dispersion, (2) fall

velocity and cohesiveness, (3) deposition on the river bottom, (4) river
bed erosion, and (5) tributary flow. Necessary data include: (1) con-
centration of each of j sediment types per unit volume of water; (2)
time after release; (3) longitudinal, lateral, and vertical velocities;
(4) particle fall velocity of j sediment types; (5) longitudinal,
lateral, and vertical diffusion coefficients of j sediment types; (6)
river width; (7) flow depth; (8) sediment deposition and erosion rates
for j sediment types; and (9) estimates of several coefficients dealing
with erodibility, critical shear stress, and impaction probability on
the river bottom for each of j sediment types.

Basically, the same data are required for the particulate con-
taminant transport submodel. However, this submodel also takes into
account the adsorption and desorption of a dissolved pollutant by
sediments and removal of the pollutant by chemical or biological means.

The dissolved pollutant submodels of FETRA and SERATRA also account
for convection and dispersion of pollutant, adsorption and desorption of
pollutant from water, chemical and biological decay of pollutant, and
the effects of tributaries. In addition to data required for the two
previous submodels, the dissolved contaminant model needs information
about the following: (1) distribution coefficient between dissolved and
particulate contaminant adsorbed on each of the j sediment types; (2)
amount of particulate pollutant per unit mass of j sediments; (3) mass
of contaminant dissolved per unit volume of water; (4) longitudinal,
lateral, and vertical diffusion coefficients for dissolved contaminant;
and (5) chemical and biological decay rate of contaminant.

As the reader has likely noticed, the amount of input data needed
for the FETRA and SERATRA models is great. Given the diversity of input

data and the difficulty in measuring some of the input parameters, it is doubtful that either FETRA or SERATRA would be used for a one-time assessment of a pollutant release into a river, such as an accidental release unless prior input data had previously been collected. However, if the model input data were available for a particular site, the models would be well suited to estimate pollutant transfer downstream following some accidental release. We suggest FETRA and SERATRA for those situations in which pollutant sorption to sediments may be important or when the pollutant is a particulate.

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5. AQUATIC FOOD CHAIN TRANSPORT

Roberta W. Shor

5.1 The Basic Model

Models that describe the transport of radionuclides from liquid effluents to man via aquatic foods generally take the form[1-3]

$$R_{aijp} = U_{ap} C_i B_{ip} D_{iaj} \quad ,$$

where

R_{aijp} = dose to organ j of an individual of age group a due to nuclide i via aquatic food p (millirem/year),

U_{ap} = ingestion rate of aquatic food p by an individual of age a (kg/year),

C_i = concentration of radionuclide i in water (pCi/liter),

B_{ip} = the equilibrium ratio of the concentration of radionuclide i in aquatic food p to its concentration in water (liter/kg),

D_{aipj} = dose conversion factor for age group a, radionuclide i, aquatic food p, and organ j (millirem/pCi).

Methods of calculating C_i are considered in Sect. 4 of this report; U_{ap} is considered in Sect. 6, and D_{aipj} is considered in Sect. 7. The bioaccumulation factor B_{ip} is considered in the remainder of this section.

5.2 The Bioaccumulation Factor

The transfer of radionuclides from contaminated water to human food through various tropic levels of aquatic life is represented by a single parameter, the bioaccumulation factor, B_{ip}. Its value is usually tabulated by nuclide for several tropic levels in freshwater and seawater.[4-7]

Reported values for B_{ip} vary commonly by an order of magnitude—in some cases by three and four orders of magnitude. Examples of the larger range of values are cesium in freshwater fish and managese in both freshwater and marine malluses.[5-7]

The accumulation of radionuclides in aquatic food occurs by complex interactions of biological, chemical and physical factors including some that can confound the assumption of equilibrium in the definition of B_{ip} (see 5.1). In addition, although nuclides concentrate in different tissues, the tabulations do not always designate the tissue. When different B_{ip}'s of an organism are known, the values given here refer to edible parts.[5] Stable elements and their congences markedly affect the accumulation of radionuclide in piota. Extensive studies of ^{89}Sr, ^{90}Sr, (and ^{137}Cs) have shown their ecological accumulation to be usually inverse functions of congenes stable element concentration, of calcium (and potassium) in both aquatic and terrestial biota.[8-11]

The B_{ip}'s for Cs and Sr in freshwater fish have been correlated from data from a number of studies in natural waters by Vanderploeg et al.[7] B_{ip} values may be estimated from site-specific concentrations of potassium and calcium in water according to the following:

1. <50 ppm sediment in unfiltered water

   piscivorous fish, $B_{ip}(Cs) = 1.5 \times 10^4/[K]_w$

   nonpiscivorous fish, $B_{ip}(Cs) = 5 \times 10^3/[K]_w$

2. >50 ppm sediment in unfiltered water

   piscivorous fish, $B_{ip}(Cs) = 3 \times 10^4/[K]_w$

   nonpiscivorous fish, $B_{ip}(Cs) = 1 \times 10^3/[K]_w$

3. fish muscle

$$B_{ip}(Sr) = \exp \{5.18 \pm 1.11 \text{ (s.e.)} - 1.21 \pm 0.37 \text{ (s.e.)} \ln [Ca]_w\} ,$$

where

$[K]_w$ = equilibrium concentration of K in the water (ppm),

$[Ca]_w$ = equilibrium concentration of Ca in the water (ppm), and

s.e. = standard error of the mean.

Site-specific information should be used in estimating bioaccumulation factors, especially for cesium in freshwater environments, if precision of one order of magnitude is desired.

In the absence of site-specific information, values of B_{ip} for different nuclides have been tabulated. Table 5.1 presents the results of an analysis[12] of B_{ip} values for strontium, iodine, and cesium in freshwater finfish taken from Vanderploeg et al.[7] Values of B_{ip} currently used at Oak Ridge National Laboratory [13] for both freshwater and marine species are shown in Table 5.2. The tabulation consists of selected values from Thompson et al.[5] Most of these values have been calculated from separate analyses of the stable elements in biota and water that may not have originated in the same part of the world. Although these values are reasonable as shown by comparison with experiments in which equilibrium between water and biota was attempted or attained, site specific values may possibly exhibit wide variations from them.

Accidental releases have not been judged important in aqueous food pathway assessments because of the availability of time for counter measures to be taken. A study was made of the accidental consequences of a nuclear submarine collision with a surface vessel in a harbor. The

authors conclude that a single meal of fish or shell fish consumed before the food supply was monitored would result in exposure within the range of "acceptable" dose levels.[14]

Table 5.1 Results of a statistical analysis of B_{ip}(liters/kg) for Sr, I, and Cs in freshwater finfish[a]

Element	μ	σ	X̄
Sr	2.4	1.8	60
I	3.5	0.61	41
Cs	7.2	0.86	1900

[a]Hoffman, F. O. (ref. 12).

Table 5.2 Values of B_{ip}(liter/kg) for various elements in aquatic foods[a]

Element	Freshwater		Marine	
	Invertebrates	Fish	Invertebrates	Fish
Na	1.7E + 01	2.0E + 01	1.0E 00	1.0E 00
P	1.0E + 05	1.0E + 05	1.0E + 04	1.0E + 04
S	1.0E + 02	7.5E + 02	4.0E 00	4.0E 00
Mn[b]	1.0E + 05	3.2E - 01[c]	5.0E + 04	3.0E + 03
Fe	3.2E + 03	1.0E + 02	2.0E + 04	1.0E + 03
Co[b]	1.0E 04	3.0E 02	1.0E + 04	1.0E + 02
Ni	1.0E + 02	1.0E + 02	1.0E + 02	5.0E + 02
Zn	1.0E + 04	1.0E + 03	5.0E + 04	5.0E + 03
Sr[b]	3.0E + 02		1.0E 00	1.0E 00
Y	1.0E + 03	2.5E + 01	1.0E + 02	3.0E + 01
Zr	6.7E 00	3.3E 00	1.0E + 02	3.0E + 01
Nb	1.0E + 02	3.0E + 04	2.0E + 02	1.0E + 02
Tc	5.0E 00	1.5E + 01	1.0E + 02	1.0E + 01
Ru	3.0E + 02	1.0E + 01	1.0E + 02	3.0E 00
Ag	7.7E + 02	2.3E 00	5.0E + 03	1.0E + 03
Sb	1.0E + 01	1.0E 00	1.0E + 03	1.0E + 03
Te	6.1E + 03	4.0E + 02	1.0E + 02	1.0E + 01
I[b]	4.0E + 02		1.0E + 02	2.0E + 01
Cs[b]	1.0E + 03		5.0E + 01	3.0E + 01
Ce	1.0E + 03	2.5E + 01	1.0E + 02	3.0E + 01
Pm	1.0E + 03	2.5E + 01	1.0E + 03	1.0E + 02
Pb	1.0E + 02	3.0E + 02	1.0E + 03	3.0E + 02
Bi	1.0E + 05	1.5E + 01	1.0E + 05	1.5E + 01
Po	2.0E + 04	5.0E + 01	2.0E + 04	2.0E + 03
Ra	2.5E + 02	5.0E + 01	1.0E + 02	5.0E + 01
Th	5.0E + 02	3.0E + 01	2.0E + 03	1.0E + 04
Pa	1.1E + 02	1.1E + 01	1.0E + 01	1.0E + 01
U	1.0E + 02	1.0E + 01	1.0E + 01	1.0E + 01
Np	4.0E + 02	1.0E + 01	1.0E + 01	1.0E + 01
Pu	1.0E + 02	3.5E + 02	1.0E + 02	3.5E 00
Am	1.0E + 03	2.5E + 01	1.0E + 03	2.5E + 01
Cm	1.0E + 03	2.5E + 01	1.0E + 03	2.5E + 01

[a]Values taken from ref. 7 unless otherwise indicated.
[b]Values taken from ref. 5 for freshwater; fish muscle.
[c]Divided by $[Mn]_w$ water concentration of Mn (ppm).

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6. ADULT DIETARY INTAKE AND INHALATION RATES

Elizabeth M. Rupp

The internal radiation dose to man as a result of exposure to contaminated air, water, and/or food is determined in part by the amount of material which is incorporated into the human body via food and inhalation pathways. Dietary parameters and inhalation rates considered in this study are as follows:

1. the consumption of milk, U^M_{ap} (liter/year);

2. the consumption of water, U^W_{ap} (liter/year);

3. the consumption of fruits, leafy vegetables, below-ground vegetables, other vegetables, grain, beef, other meats and poultry, finfish, and shellfish, U^F_{ap} (kg/year);

4. the volumetric inhalation rate, U^A_{ap} (m^3/year).

Table 6.1 presents the annual average values of ingestion and inhalation rates for adult individuals. While these factors are undoubtedly related to age and individual variability, only adult values are given in Table 6.1 because the internal dose conversion factors provided in Sect. 7 are only for adult individuals. A more detailed analysis of human dietary intake and inhalation rates is given elsewhere.[1]

Table 6.1. Estimated values of ingestion and inhalation
rates for adult individuals

Item	Average value	Reference
--kg/year--		
Fruits	68	2
Leafy vegetables	18	3
Below ground vegetables	28	2
Other vegetables	45	2
Grains (flour equivalent)	35	2
Beef	32	2
Other meats, poultry	63	2
Finfish	4.4	8, 9
Shellfish	1.3	8, 9
--liters/year--		
Milk[a]	112	2
Tap water	93	10
Other beverages	400	10
--m^3/year--		
Inhalation rate	8030	11

[a]Include all milk drinks, fresh cream, ice cream, and
small amounts of cheese.

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7. DOSE CONVERSION FACTORS

H. Robert Meyer
Donald E. Dunning, Jr.
David C. Kocher
Kathy K. Kanak

7.1 Introduction

The radiological impact to man resulting from operation of a nuclear-related facility may be assessed by calculating the dose to individuals and populations residing near the facility. There are a number of potential modes of exposure to man from radioactive effluents released to the environment by breeder reactor facilities. Modes contributing to the vast majority of human exposure are inhalation, ingestion, immersion in contaminated air, and exposure to contaminated ground surfaces. Dose conversion factors are used to estimate the dose resulting from these exposures.

7.2 Internal Dose Conversion Factors

Table 7.1 lists recommended dose conversion factors for inhalation and ingestion of certain radionuclides determined to be of potential significance in breeder reactor fuel cycles. These factors are taken from recently prepared documents by Killough et al.[1] and by Dunning et al.,[2] and were calculated by the use of dosimetric criteria provided by the International Commission on Radiological Protection[3] and other recognized authorities. These factors are computed with the computer code INREM II, which approximates the solutions of differential equations which model intake, translocation and metabolism of a radionuclide and its progeny.

Table 7.1 Fifty-year dose commitments for inhalation or ingestion of certain radionuclides (rem/μCi intake)[a]

Nuclide	Pathway	Solubility class[b]	Lungs	Total body	Ovaries	Total endosteal cells	Testes	Uptake Fraction (SI:B)[c]
^{3}H	Inhalation		1.25E-4	1.25E-4	1.24E-4	9.85E-5	1.25E-4	0.95
	Ingestion		8.36E-5	8.30E-5	8.29E-5	6.56E-5	8.30E-5	0.95
^{14}C	Inhalation		6.18E-6	1.41E-5	5.29E-6	5.08E-5	5.42E-6	0.95
	Ingestion		8.94E-4	1.92E-3	7.36E-4	7.05E-3	7.23E-4	0.95
^{22}Na	Inhalation		9.43E-3	9.16E-3	1.02E-2	1.60E-2	7.46E-3	0.95
	Ingestion		9.13E-3	1.36E-2	1.57E-2	2.41E-2	1.13E-2	0.95
^{54}Mn	Inhalation	D	4.18E-3	4.10E-3	3.79E-3	5.86E-3	2.25E-3	0.1
		W	2.47E-2	3.66E-3	2.81E-3	3.44E-3	9.07E-4	0.1
	Ingestion		8.03E-4	1.23E-3	3.61E-3	1.37E-3	7.10E-4	0.1
^{55}Fe	Inhalation	W	3.20E-3	6.29E-4	1.11E-3	7.40E-4	1.05E-3	0.1
		Y	2.51E-2	6.09E-4	5.16E-4	3.43E-4	4.86E-4	0.1
	Ingestion		1.23E-5	3.50E-4	6.59E-4	4.37E-4	6.20E-4	0.1
^{58}Co	Inhalation	Y	5.92E-2	4.17E-3	2.36E-3	2.58E-3	3.64E-4	0.05
		W	2.96E-2	2.65E-3	2.60E-3	1.78E-3	6.89E-4	0.05
	Ingestion		2.64E-4	7.65E-4	3.92E-3	4.58E-4	5.39E-4	0.05
^{60}Co	Inhalation	W	0.132	1.71E-2	1.70E-2	1.47E-2	9.51E-3	0.05
		Y	1.27	8.20E-2	1.85E-2	5.08E-2	6.12E-3	0.05
	Ingestion		2.68E-3	4.37E-3	1.24E-2	3.99E-3	3.82E-3	0.05
^{65}Zn	Inhalation	D	1.18E-2	1.40E-2	1.74E-2	1.77E-2	1.28E-2	0.5
		W	2.54E-2	1.05E-2	1.18E-2	1.23E-2	8.30E-3	0.5
		Y	7.65E-2	1.24E-2	9.50E-3	1.21E-2	6.31E-3	0.5
	Ingestion		1.02E-2	1.26E-2	1.65E-2	1.58E-2	1.15E-2	0.5
^{89}Sr	Inhalation	D	6.55E-3	3.89E-3	2.18E-3	2.74E-2	2.18E-3	0.2
		Y	0.313	4.76E-3	4.35E-5	5.44E-4	4.28E-5	0.01

Table 7.1 (continued)

	Pathway	Solubility class	Lungs	Total body	Ovaries	Total endosteal cells	Testes	Uptake Fraction (SI:B)
^{89}Sr	Ingestion		9.64E-8	1.80E-3	8.56E-4	1.08E-2	8.55E-4	0.2
			1.58E-8	4.87E-4	4.33E-5	5.38E-4	4.28E-5	0.01
^{90}Sr	Inhalation	D	9.89E-3	0.241	1.53E-2	2.20	1.53E-2	0.2
		Y	8.49	0.152	3.65E-4	0.233	3.65E-3	0.01
	Ingestion		5.94E-9	9.45E-2	5.99E-3	0.859	5.99E-3	0.2
			2.97E-10	5.02E-3	3.00E-4	4.30E-2	3.00E-4	0.01
^{90}Y	Inhalation		3.93E-2	8.90E-4	9.07E-5	9.32E-4	9.07E-5	1.0E-4
	Ingestion		9.80E-9	5.07E-4	1.25E-7	1.22E-6	1.18E-7	1.0E-4
^{91}Y	Inhalation	W	0.198	5.67E-3	7.28E-4	1.79E-2	7.21E-4	1.0E-4
	Ingestion		3.30E-7	4.37E-4	1.34E-5	1.97E-5	1.86E-6	1.0E-4
^{95}Zr	Inhalation	W	6.86E-2	4.95E-3	3.14E-3	8.99E-3	1.18E-3	2.0E-3
		Y	0.131	5.56E-3	2.31E-3	3.24E-3	3.29E-4	2.0E-3
	Ingestion		8.29E-5	5.49E-4	3.02E-3	3.55E-4	2.99E-4	2.0E-3
^{95}Nb	Inhalation		3.10E-2	1.94E-3	1.64E-3	1.19E-3	2.46E-4	1.0E-2
	Ingestion		9.81E-5	5.04E-4	2.99E-3	3.19E-4	3.58E-4	1.0E-2
^{99}Tc	Inhalation	D	9.63E-4	1.69E-4	2.41E-4	3.11E-4	2.41E-4	0.80
		W	5.22E-2	8.87E-4	2.12E-4	2.75E-5	2.12E-4	0.80
	Ingestion		0.0	2.15E-4	3.17E-4	4.10E-4	3.17E-4	0.80
^{103}Ru	Inhalation	Y	5.84E-2	1.98E-3	1.20E-3	9.25E-4	2.55E-4	4.0E-2
	Ingestion		1.36E-4	5.29E-4	2.17E-3	3.96E-4	4.27E-4	4.0E-2
^{106}Ru	Inhalation	Y	3.85	6.18E-2	7.67E-3	1.00E-2	6.97E-3	4.0E-2
	Ingestion		2.17E-4	5.94E-3	8.96E-3	9.58E-3	8.14E-3	4.0E-2

Table 7.1 (continued)

	Pathway	Solubility class	Target Lungs	Target Total body	Target Ovaries	Target Total endosteal cells	Target Testes	Uptake Fraction (SI:B)
^{127}Te	Inhalation	W	2.48E-3	5.13E-5	5.29E-3	3.06E-5	5.22E-5	0.25
	Ingestion							
^{127m}Te	Inhalation	W	0.118	2.42E-3	2.53E-3	2.91E-3	2.48E-3	0.25
	Ingestion		2.32E-5	7.60E-4	2.64E-3	3.08E-3	2.55E-3	0.25
^{129m}Te	Inhalation	W	0.153	3.20E-3	4.15E-3	3.18E-3	4.02E-3	0.25
	Ingestion		7.19E-5	1.29E-3	5.23E-3	3.83E-3	4.96E-3	0.25
^{129}I	Inhalation	D	7.88E-4	2.05E-3 Thyroid: 4.97	3.78E-4	5.65E-4	3.57E-4	0.95
	Ingestion		1.80E-4	3.19E-3 Thyroid: 7.78	5.92E-4	8.79E-4	5.58E-4	0.95
^{131}I	Inhalation	D	2.40E-3	6.13E-4 Thyroid: 1.13	3.98E-5	2.21E-4	2.22E-5	0.95
	Ingestion		2.86E-4	9.08E-4 Thyroid: 1.81	6.11E-5	3.32E-4	3.53E-5	
^{134}Cs	Inhalation	D	3.38E-2	4.55E-2	6.46E-2	5.89E-2	5.13E-2	0.95
	Ingestion		4.68E-2	6.84E-2	9.74E-2	8.86E-2	7.73E-2	0.95
^{137}Cs	Inhalation	D	1.62E-2	3.26E-2	5.00E-2	5.31E-2	4.44E-2	0.95
	Ingestion		1.99E-2	4.91E-2	7.54E-2	7.99E-2	6.68E-2	0.95
^{144}Ce	Inhalation	Y	2.92	4.68E-2	6.29E-4	1.47E-2	4.58E-4	1.0E-4
		W	0.678	5.11E-2	4.65E-3	0.142	4.40E-3	1.0E-4
		D	1.73E-2	0.175	1.94E-2	0.610	1.88E-2	1.0E-4
	Ingestion		4.48E-6	1.01E-3	2.42E-4	1.47E-4	2.29E-5	1.0E-4
^{147}Pm	Inhalation	Y	0.287	5.30E-3	1.24E-4	4.33E-3	1.23E-4	1.0E-4
		W	3.63E-2	6.58E-3	6.34E-4	2.23E-2	6.34E-4	1.0E-4
		D	7.06E-4	2.46E-2	2.58E-3	9.07E-2	2.58E-3	1.0E-4
	Ingestion		4.37E-10	5.02E-5	5.55E-7	1.89E-5	5.39E-7	1.0E-4

Table 7.1 (continued)

	Pathway	Solubility class	Target					Uptake Fraction (SI:B)
			Lungs	Total body	Ovaries	Total endosteal cells	Testes	
210Pb	Inhalation	W	6.18	15.2	0.669	19.5	0.669	0.08
	Ingestion		6.92E-5	7.44	0.302	9.63	0.302	0.08
211Pb	Inhalation	W	0.140	2.20E-3	3.12E-5	3.88E-3	3.12E-5	0.08
	Ingestion		6.75E-7	5.77E-5	8.42E-6	5.28E-4	5.42E-6	0.08
212Pb	Inhalation	W	1.79	3.65E-2	1.97E-3	0.166	1.71E-3	0.08
	Ingestion		5.33E-5	7.75E-3	1.87E-3	0.119	6.32E-4	0.08
214Pb	Inhalation	W	0.116	1.96E-3	1.09E-4	2.82E-3	1.08E-4	0.08
	Ingestion		7.59E-6	8.14E-5	5.11E-5	3.03E-4	1.32E-5	0.08
210Bi	Inhalation	W	1.02	2.79E-2	8.68E-3	4.05E-3	8.68E-3	0.05
	Ingestion		5.20E-11	3.04E-3	2.32E-3	5.89E-4	1.25E-3	0.05
212Bi	Inhalation	W	0.271	4.32E-3	3.17E-4	2.05E-3	3.15E-4	0.05
	Ingestion		1.29E-5	1.39E-4	1.15E-4	3.03E-4	5.32E-5	0.05
213Bi	Inhalation	W	9.77E-3	1.68E-4	1.62E-5	4.29E-5	1.26E-5	0.05
	Ingestion		1.79E-5	5.27E-5	7.68E-5	2.03E-5	1.14E-5	0.05
214Bi	Inhalation	W	9.39E-2	1.49E-3	9.13E-5	5.51E-4	9.06E-5	0.05
	Ingestion		6.36E-6	4.00E-5	2.12E-5	4.16E-5	5.82E-6	0.05
210Po	Inhalation	W	45.8	1.56	0.803	0.374	0.803	0.1
	Inhalation	D	1.03	2.80	2.58	1.20	2.58	0.1
	Ingestion		1.14E-8	0.562	0.521	0.242	0.521	0.1
220Rn	Inhalation	gas	2.16E-3	3.22E-5	1.21E-7	1.90E-5	9.60E-8	0
	Ingestion		8.47E-8	2.22E-5	3.12E-6	3.67E-4	1.31E-6	0
222Rn	Inhalation	gas	2.34E-3	3.46E-5	2.72E-7	1.03E-5	2.66E-7	0
	Ingestion		1.15E-4	4.59E-3	5.40E-3	1.31E-2	7.06E-4	0
223Ra	Inhalation	W	46.4	1.12	0.221	7.50	0.221	0.2
	Ingestion		5.14E-5	0.561	0.272	9.16	0.272	0.2
224Ra	Inhalation	W	8.75	0.267	6.11E-2	2.47	6.03E-2	0.2
	Ingestion		9.21E-5	0.189	8.27E-2	3.30	8.05E-2	0.2

Table 7.1 (continued)

	Pathway	Solubility class	Lungs	Total body	Ovaries	Total endosteal cells	Testes	Uptake Fraction (SI:B)
					Target			
225Ra	Inhalation	W	28.8	0.942	0.243	8.75	0.226	0.2
	Ingestion		3.37E-4	0.517	0.260	8.72	0.259	0.2
226Ra	Inhalation	W	56	19.5	0.677	227	0.676	0.2
	Ingestion		1.76E-3	15.8	0.590	202	0.590	0.2
228Ra	Inhalation	W	4.81	11	1.1	262	1.1	0.2
	Ingestion		2.23E-3	8.4	0.778	203	0.778	0.2
225Ac	Inhalation	Y	49.3	0.783	4.5E-2	1.16	3.72E-2	1.0E-3
		W	42.5	1.54	0.25E-2	16.9	0.25E-2	1.0E-3
	Ingestion		2.01E-4	3.35E-2	3.06E-2	0.364	2.40E-2	1.0E-3
227Ac	Inhalation	Y	1030	255	35.	4470	2.40E-2	1.0E-3
		W	84.1	581	93.	1.22E+4	92.	1.0E-3
	Ingestion		3.53E-4	4.8	0.77	101	0.77	1.0E-3
227Th	Inhalation	Y	68.7	1.18	5.93E-2	3.14	5.90E-2	1.0E-3
		W	52.7	2.18	0.25	28.6	0.25	1.0E-3
	Ingestion		1.19E-5	2.40E-2	6.57E-3	0.45	6.02E-3	1.0E-3
228Th	Inhalation	Y	716	28.6	2.2	287	2.1	1.0E-3
		W	117	71.7	10.	1590	10	1.0E-3
	Ingestion		6.43E-5	0.602	9.01E-2	13.7	8.96E-2	1.0E-3
229Th	Inhalation	Y	1230	387	59.	8400	59	1.0E-3
		W	116	830	140.	2.06E+4	140	1.0E-3
	Ingestion		2.00E-2	6.82	12	170	1.2	1.0E-3
230Th	Inhalation	Y	526	155	37	4280	37.	1.0E-3
		W	54.2	347	91.	1.06E+4	91.	1.0E-3
	Ingestion		4.62E-6	2.85	0.75	87.4	0.75	1.0E-3
231Th	Inhalation	Y	4.16E-3	1.21E-4	3.39E-5	7.01E-4	7.18E-6	1.0E-3
		W	3.91E-3	1.77E-4	4.96E-5	1.98E-3	2.69E-5	1.0E-3
	Ingestion		6.41E-7	8.07E-5	8.26E-5	2.28E-5	2.69E-6	1.0E-3

Table 7.1 (continued)

				Target				Uptake Fraction (SI:B)
	Pathway	Solubility class	Lungs	Total body	Ovaries	Total endosteal cells	Testes	
^{232}Th	Inhalation	Y	454	158	32	4870	32	1.0E-3
		W	46.5	354	79	1.19E+4	79	1.0E-3
	Ingestion		1.04E-4	2.90	0.65	98.5	0.65	1.0E-3
^{234}Th	Inhalation	Y	0.243	3.92E-3	1.31E-4	5.18E-4	8.27E-5	1.0E-3
		W	0.178	4.70E-3	1.39E-3	8.23E-3	1.34E-3	1.0E-3
	Ingestion		1.80E-6	6.56E-4	1.15E-4	1.05E-4	2.30E-5	1.0E-3
^{231}Pa	Inhalation	Y	594	413	78	9370	78	1.0E-3
		W	57.6	890	181	2.17E+4	181	1.0E-3
	Ingestion		4.35E-4	7.25	1.5	178	1.5	1.0E-3
^{233}Pa	Inhalation	Y	6.22E-2	1.32E-3	4.89E-4	4.19E-4	7.01E-5	1.0E-3
		W	4.43E-2	1.58E-3	8.53E-4	2.47E-3	4.44E-4	1.0E-3
	Ingestion		1.45E-5	2.61E-4	8.98E-4	9.41E-5	7.80E-5	1.0E-3
^{234}Pa	Inhalation	Y	5.23E-3	2.10E-4	2.44E-4	8.51E-5	1.93E-5	1.0E-3
		W	4.89E-3	2.11E-4	2.15E-4	1.34E-4	2.81E-5	1.0E-3
	Ingestion		5.82E-5	3.13E-4	1.26E-3	1.03E-4	9.00E-5	1.0E-3
^{232}U	Inhalation	Y	887	80	6.4	733	6.4	2.0E-3
		W	66.5	39	8.5	649	8.5	5.0E-2
		D	1.14	119	27.4	2030	27.5	5.0E-2
	Ingestion		9.52E-5	0.444	0.112	8.3	0.113	2.0E-1
			2.38E-3	11.1	2.81	208	2.82	5.0E-1
^{233}U	Inhalation	Y	542	19.5	1.17	3.59	1.17	2.0E-3
		W	55.9	10.9	3.40	10.4	3.40	5.0E-2
		D	0.950	33.5	11.3	34.6	11.3	5.0E-2
	Ingestion		6.53E-5	0.138	4.64E-2	0.142	4.64E-2	2.0E-3
			1.63E-3	3.43	1.16	3.55	1.16	5.0E-2
^{234}U	Inhalation	Y	536	19.3	1.16	353	1.16	2.0E-3
		W	55.4	10.8	3.37	10.2	3.37	5.0E-2
		D	0.941	33.2	11.2	34.0	11.2	5.0E-2
	Ingestion		6.46E-5	0.137	4.59E-2	0.139	4.59E-2	2.0E-3
			1.61E-3	3.40	1.15	3.48	1.15	5.0E-2

Table 7.1 (continued)

| | Pathway | Solubility class | Target | | | | | Uptake Fraction (SI:B) |
			Lungs	Total body	Ovaries	Total endosteal cells	Testes	
^{235}U	Inhalation	Y	484	17.4	1.05	3.37	1.06	2.0E-3
		W	49.9	9.77	3.04	8.74	3.03	5.0E-2
		D	0.854	29.9	10.1	29.0	10.1	5.0E-2
	Ingestion		9.27E-5	0.123	4.24E-2	0.119	4.14E-2	2.0E-3
			2.08E-3	3.07	1.04	2.97	1.03	5.0E-2
^{236}U	Inhalation	Y	506	18.2	1.09	3.18	1.09	2.0E-3
		W	52.2	10.2	3.17	9.22	3.17	5.0E-2
		D	0.887	31.3	10.5	30.7	10.5	5.0E-2
	Ingestion		6.09E-5	0.129	4.32E-2	0.126	4.32E-2	2.0E-3
			1.52E-3	3.20	1.08	3.14	1.08	5.0E-2
^{238}U	Inhalation	Y	480	17.1	1.03	2.89	1.04	2.0E-3
		W	49.0	9.57	2.99	8.36	3.01	5.0E-2
		D	0.834	29.3	9.96	27.8	10.0	5.0E-2
	Ingestion		6.01E-5	0.121	4.08E-2	0.114	4.10E-2	2.0E-3
			1.50E-3	3.00	1.02	2.85	1.02	5.0E-2
^{237}Np	Inhalation	Y	569	167	40	4240	40.	1.0E-3
		W	58.6	374	98.	1.05E-4	99.	1.0E-3
	Ingestion		8.73E-4	3.08	0.81	86.8	0.81	1.0E-3
^{238}Pu	Inhalation	Y	608	149	35.	3270	35.	3.0E-5
		W	63.6	340	89.	8400	89	3.0E-5
	Ingestion		1.14E-7	8.51E-2	2.21E-2	2.09	2.21E-2	3.0E-5
^{239}Pu	Inhalation	Y	580	169	40.	4160	40.	3.0E-5
		W	59.8	381	100.	1.04E+4	100.	3.0E-5
	Ingestion		9.35E-8	9.51E-2	2.48E-2	2.58	2.47E-2	3.0E-5
^{240}Pu	Inhalation	Y	579	169	40.	4150	40	3.0E-5
		W	59.8	380	100.	1.04E+4	100.	3.0E-5
	Ingestion		1.16E-7	9.49E-2	2.48E-2	2.58	2.47E-2	3.0E-5

Table 7.1 (continued)

	Pathway	Solubility class	Target					
			Lungs	Total body	Ovaries	Total endosteal cells	Testes	Uptake Fraction (SI:B)
241Pu	Inhalation	Y	1.10	3.52	9.00E-1	85.4	0.90	3.0E-5
		W	1.54E-2	7.75	2.0	194	2.1	3.0E-5
	Ingestion		7.18E-8	1.93E-3	5.10E-4	4.83E-2	5.11E-4	3.0E-5
242Pu	Inhalation	Y	550	161	38	4190	38.	3.0E-5
		W	56.8	361	95.	1.05E+4	95	3.0E-5
	Ingestion		1.91E-7	9.02E-2	2.36E-2	2.60	2.35E-2	3.0E-5
241Am	Inhalation	Y	615	175	41	3900	41.	1.0E-3
		W	63.6	392	103.	9730	103.	1.0E-3
	Ingestion		1.28E-4	3.22	0.85	80.2	0.85	1.0E-3
243Am	Inhalation	Y	595	175	41.	4170	42.	1.0E-3
		W	61.4	391	103.	1.03E+4	103.	1.0E-3
	Ingestion		9.56E-4	3.21	0.85	85.2	0.85	1.0E-3
242Cm	Inhalation	Y	172	3.67	2.70E-1	22.5	0.26	1.0E-3
		W	54.8	8.76	2.0	168	2.0	1.0E-3
	Ingestion		2.65E-6	7.34E-2	1.85E-2	1.53	1.85E-2	1.0E-3
243Cm	Inhalation	Y	627	113	25.	2150	25.	1.0E-3
		W	67.5	261	68.	5780	68.	1.0E-3
	Ingestion		3.26E-4	2.15	0.17	47.8	0.56	1.0E-3
244Cm	Inhalation	Y	607	86.5	19.	1620	19.	1.0E-3
		W	66.8	204	53.	4560	53.	1.0E-3
	Ingestion		3.54E-6	1.68	0.44	37.7	0.43	1.0E-3

[a] D = daily, W = weekly, Y = yearly (ref. 3).

[b] SI:B = Gastrointestinal uptake fraction; small intestine: blood (refs. 1,2).

[c] Activity median aerodynamic diameter (AMAD) = 1 μm (ref. 1).

The INREM II computer code utilizes an adaptation of the Task Group Lung Model[3] to describe radionuclide deposition and retention dynamics within the respiratory tract. A catenary gastrointestional (GI) tract model based on the transit times recommended by Eve[4] simulates retention in the GI tract. Retention in other organs is represented by multicompartment models consisting of series of decaying exponential terms. Detailed discussion of these models and assumptions is available in references 1 and 2.

In the context of radionuclides released during routine breeder reactor fuel cycle operations, it is useful to consider the calculation of 50-year dose commitments in terms of three radionuclide classes, based on effective half-life of radionuclides in the human body. Depending on the organ under consideration, one of several approaches will apply.

For radionuclides that have short biological half-times of residence or short radiological half-lives, the 50-year dose commitment would be approximately the same as the annual dose. For example, all of the dose commitment for ^{220}Rn exposure (including radon daughters) is received during the first year.

For radionuclides of intermediate longevity in humans, dose would be accrued at a declining rate over the entire 50-year period, based on models and associated parameters specific to the organ and nuclide of concern. For example, ^{232}U (physical $T_{1/2}$ = 71.7 years) is retained in bone according to the uranium retention function:

$$R_{(bone)}(t) = 0.90 \, e^{-0.693 \, t/20} + 0.10 \, e^{-0.693 \, t/5000},$$

where t is the time of residence in days. This expression indicates that the initial component of retained uranium is removed rapidly in biological processes with a half-time of 20 days. The second component in the expression contributes the majority of the total 50-year dose commitment from ^{232}U, at a rate which decreases with a half-time of 13.7 years.

For elements which are eliminated very slowly from the body, such as plutonium, and which possess very long physical half-lives [$T_{1/2}$ (^{239}Pu) = 2.4 x 10^4 years)], an individual will continue to accrue dose after intake of the radioisotope at a relatively constant rate for the entire 50-year period of interest. Under such conditions, the approximate dose received during the year after the radionuclide enters the body is obtained by dividing the dose commitment by 50. Thus, the approximate average annual dose rate is only 1/50 of the dose commitment. If an individual is exposed to effluents for a 20-year operating life of the plant, his annual dose rate during the twentieth year is about 20 times the annual dose rate from one year of exposure.

These generalized observations are approximately correct for the conditions cited. However, a detailed calculation must be made to determine the actual dose received in a given year.

Table 7.1 lists 50-year dose commitments for inhalation and ingestion of those radionuclides of interest in this study. These dose equivalents take into account the contributions of radioactive daughters formed by decay of a parent within the body. Dissimilar migrations of daughter elements are also considered. Where a choice of solubility class exists,

based on a choice of the chemical form of a particular released radio-nuclide, dose conversion factors for all solubility classes applicable are listed. The reader is referred to Killough et al.,[1] Dunning et al.,[2] or Morrow et al.[3] for detailed discussions of this subject. A quality factor of 20 is used for alpha particles, as suggested by the ICRP.[5]

While dose conversion factors have been calculated elsewhere for 22 target organs,[1,2] the purposes of this report may be best served by focusing on five organs of major interest from the standpoint of health risk assessment. These organs are lungs, total body, ovaries, testes, and total endosteal cells (sensitive cell layer in bone). Dose conversion factors for lung are calculated as dose to the entire lung; this is commonly identified as the "smeared lung" dose. Gastrointestinal uptake fractions (small intestine: blood) are listed for radionuclides in the final column of Table 7.1. The reader is referred to Killough et al.[1] and Dunning et al.[2] for detailed discussion of this subject.

7.3 External Dose Conversion Factors

Tables 7.2 and 7.3 list external dose conversion factors for the radionuclides of interest as discussed above. Dose conversion factors for immersion in contaminated air are listed for: (1) β dose rate at the skin surface of an exposed body (body surface), and (2) photon (γ and X-ray) dose rate to various body organs (lungs, ovaries, skeleton, testes, and total body). Similarly, dose conversion factors for exposure to radionuclides deposited on contaminated ground surfaces are calculated for body surface and for the five organs, as above. These dose conversion factors are from the report by Kocher.[6]

Table 7.2 External dose conversion factors for immersion in
contaminated air (millirem/yr/μCi/cm^3)

	Body surface (β dose rate)	Lungs	Ovaries	Skeleton	Testes	Total body
		(γ and x-ray dose rate)				
^{3}H	5.97E7	0.0	0.0	0.0	0.0	0.0
^{14}C	4.46E8	0.0	0.0	0.0	0.0	0.0
^{22}Na	1.74E9	1.17E10	7.68E9	1.40E10	1.16E10	1.25E10
^{24}Na	4.93E9	2.38E10	7.68E9	1.40E10	1.16E10	1.25E10
^{41}Ar	4.14E9	7.09E9	5.96E9	7.97E9	6.00E9	7.50E9
^{54}Mn	3.49E7	4.25E9	2.41E9	5.14E9	4.35E9	4.51E9
^{55}Fe	3.66E7	6.28E3	3.35E3	1.15E4	5.84E4	7.63E4
^{58}Co	3.01E8	4.97E9	2.66E9	6.09E9	5.24E9	5.28E9
^{60}Co	8.64E8	1.37E10	1.12E10	1.55E10	1.18E10	1.45E10
^{59}Ni	3.97E7	1.05E4	5.59E3	1.91E4	9.31E4	1.27E5
^{65}Zn	5.99E7	3.07F9	2.19E9	3.56E9	2.82E9	3.25E9
^{85}Kr	2.25E9	1.15E7	4.58E6	1.48E7	1.37E7	1.23E7
^{88}Kr	3.11E9	1.12E10	1.14E10	1.24E10	7.29E9	1.18E10
^{88}Rb	1.82E10	3.70E9	3.64E9	4.06E9	2.48E9	3.88E9
^{89}Sr	5.19E9	6.96E5	4.19E5	8.31E5	6.92E5	7.37E5
^{90}Sr	1.76E9	0.0	0.0	0.0	0.0	0.0
^{90}Y	8.25E9	6.89E1	3.97E1	3.75E2	4.64E2	5.55E2
^{91m}Y	2.39E8	2.72E9	1.15E9	3.48E9	3.17E9	2.91E9
^{91}Y	5.36E9	1.96E7	1.53E7	2.23E7	1.72E7	2.07E7
^{95}Zr	1.04E9	3.76E9	1.96E9	4.62E9	3.99E9	4.00E9
^{95}Nb	3.91E8	3.90E9	2.07E9	4.77E9	4.10E9	4.14E9
^{99}Tc	7.60E8	0.0	0.0	0.0	0.0	0.0
^{103}Ru	6.72E8	2.41E9	9.56E8	3.13E9	2.90E9	2.58E9
^{106}Ru	9.10E7	0.0	0.0	0.0	0.0	0.0
^{103m}Rh	3.36E8	2.31E5	2.14E5	1.27E6	1.67E6	1.19E6
^{106}Rh	1.25E10	1.06E9	5.00E8	1.33E9	1.19E9	1.13E9
^{110m}Ag	6.01E8	1.44E10	9.19E9	1.72E10	1.42E10	1.53E10
^{110}Ag	1.04E10	1.56E8	7.54E7	1.94E8	1.72E8	1.66E8
^{125}Sb	8.79E8	2.14E9	9.52E8	2.83E9	2.56E9	2.31E9
^{125m}Te	9.57E8	2.53E7	1.92E7	8.39E7	7.78E7	5.72E2
^{127m}Te	7.22E8	6.88E6	5.24E6	2.34E7	2.23E7	1.61E7
^{127}Te	2.00E9	2.48E7	1.09E7	3.44E7	3.13E7	2.69E7
^{129m}Te	2.40E9	1.55E8	7.93E7	2.04E8	1.79E8	1.72E8
^{129}Te	4.85E9	2.87E8	1.35E8	3.81E8	3.44E8	3.12E8
^{129}I	4.84E8	2.52E7	1.90E7	7.63E7	6.34E7	4.85E7
^{131}I	1.71E9	1.95E9	9.03E8	2.73E9	2.46E9	2.12E9
^{133m}Xe	1.71E9	1.39E8	8.20E7	2.47E8	2.17E8	1.69E8
^{133}Xe	1.22E9	1.44E8	8.56E7	3.38E8	1.85E8	1.80E8
^{135m}Xe	8.56E8	2.20E9	8.95E8	2.84E9	2.62E9	2.36E9
^{135}Xe	2.82E9	1.27E9	6.87E8	1.96E9	1.74E9	1.41E9
^{134}Cs	1.45E9	7.69E9	4.08E9	9.83E9	8.58E9	8.47E9
^{135}Cs	5.08E8	0.0	0.0	0.0	0.0	0.0
^{135}Cs	1.53E9	0.0	0.0	0.0	0.0	0.0
^{137m}Ba	5.67E8	3.04E9	1.45E9	3.79E9	3.35E9	3.24E9

Table 7.2 (continued)

	Body surface (β dose rate)	Lungs	Ovaries	Skeleton	Testes	Total body
		(γ and x-ray dose rate)				
^{144}Ce	8.22E8	8.66E7	4.83E7	1.75E8	1.10E8	1.04E8
^{144m}Pr	4.06E8	1.71E7	1.32E7	4.91E7	3.33E7	2.95E7
^{144}Pr	1.07E10	1.77E8	1.59E8	1.98E8	1.31E8	1.85E8
^{147}Pm	5.59E8	1.71E4	8.90E3	3.38E4	2.02E4	1.98E4
^{151}Sm	1.80E8	9.91E2	9.26E2	5.73E3	7.81E3	5.62E3
^{154}Eu	2.52E9	6.43E9	4.44E9	7.79E9	6.15E9	6.85E9
^{155}Eu	5.54E8	2.51E8	1.39E8	5.54E8	2.89E8	3.09E8
^{207}Tl	4.39E9	1.10E7	6.55E6	1.31E7	1.10E7	1.16E7
^{208}Tl	5.30E9	1.91E10	1.85E10	2.13E10	1.31E10	2.02E10
^{209}Tl	6.10E9	1.18E10	1.02E10	1.39E10	1.01E10	1.25E10
^{209}Pb	1.77E9	0.0	0.0	0.0	0.0	0.0
^{210}Pb	3.13E8	4.94E6	3.97E6	1.32E7	6.72E6	7.56E6
^{211}Pb	4.06E9	2.57E8	1.30E8	3.28E8	2.87E8	2.75E8
^{212}Pb	1.55E9	7.25E8	4.06E8	1.23E9	9.78E8	8.22E8
^{214}Pb	2.61E9	1.25E9	6.14E8	1.86E9	1.61E9	1.37E9
^{210}Bi	3.48E9	0.0	0.0	0.0	0.0	0.0
^{211}Bi	8.20E7	2.28E8	1.07E8	3.33E8	2.95E8	2.50E8
^{212}Bi	4.16E9	9.78E8	6.87E8	1.15E9	9.14E8	1.04E9
^{213}Bi	3.99E9	6.78E8	2.94E8	9.20E8	8.30E8	7.33E8
^{214}Bi	5.76E9	8.28E9	6.93E9	9.43E9	6.74E9	8.73E9
^{210}Po	0.0	4.33E4	2.38E4	5.26E4	4.49E4	4.60E4
^{213}Po	0.0	1.55E5	8.32E4	1.89E5	1.62E5	1.64E5
^{214}Po	0.0	5.58E5	3.04E5	6.80E5	5.81E5	5.93E5
^{217}At	0.0	1.52E6	6.14E5	2.01E6	1.86E6	1.64E6
^{219}Rn	5.18E7	2.85E8	1.38E8	4.20E8	3.73E8	3.12E8
^{220}Rn	0.0	2.82E6	1.18E6	3.61E6	3.30E6	3.02E6
^{222}Rn	0.0	2.00E6	7.96E5	2.59E6	2.40E6	2.14E6
^{221}Fr	8.18E7	1.56E8	9.06E7	2.59E8	2.19E8	1.75E8
^{223}Fr	3.33E9	2.39E8	1.43E8	1.12E9	8.06E8	7.29E8
^{223}Ra	6.43E8	6.39E8	3.33E8	1.12E9	8..06E8	7.29E8
^{224}Ra	1.84E7	5.04E7	2.80E7	8.06E7	6.99E7	5.61E7
^{225}Ra	9.42E8	2.11E7	1.65E7	5.92E7	3.63E7	3.49E7
^{226}Ra	3.11E7	3.37E7	2.00E7	5.87E7	4.68E7	3.82E7
^{228}Ra	1.14E8	1.68E-2	8.99E-3	3.08E-2	1.50E-1	2.04E-1
^{225}Ac	1.57E8	7.41E7	4.01E7	1.45E8	9.07E7	8.65E7
^{227}Ac	1.17E8	5.70E5	2.93E5	1.15E6	6.72E5	6.72E5
^{228}Ac	3.82E9	4.82E9	3.20E9	5.85E9	4.73E9	5.12E9
^{227}Th	3.44E8	5.18E8	2.81E8	8.42E8	6.98E8	5.28E8
^{228}Th	1.71E8	9.28E6	5.28E6	1.81E7	1.18E7	1.11E7
^{229}Th	9.42E8	4.05E8	2.24E8	8.15E8	4.99E8	4.81E8
^{230}Th	1.18E8	1.66E6	1.04E6	3.54E6	2.14E6	2.18E6
^{231}Th	1.47E9	6.09E7	3.44E7	1.37E8	7.73E7	8.00E7
^{232}Th	9.08E7	6.72E5	4.31E5	1.54E6	8.78E5	9.85E5
^{234}Th	5.33E8	3.49E7	1.97E7	7.79E7	3.92E7	4.32E7
^{231}Pa	3.40E8	1.50E8	7.64E7	2.36E8	2.02E8	1.69E8
^{233}Pa	1.70E9	9.51E8	4.67E8	1.51E9	1.22E9	1.06E9
^{234m}Pa	7.30E9	5.78E7	3.55E7	7.05E7	5.71E7	6.15E7
^{234}Pa	4.07E9	1.02E10	6.31E9	1.28E10	1.03E10	1.09E10

Table 7.2 (continued)

	Body surface (β dose rate)	Lungs	Ovaries	Skeleton	Testes	Total body
		(γ and x-ray dose rate)				
^{232}U	1.36E8	1.11E6	6.84E5	2.38E6	1.52E6	1.60E6
^{233}U	4.24E7	1.43E6	7.23E5	2.93E6	1.71E6	1.79E6
^{234}U	1.02E8	4.68E5	2.96E5	1.08E6	6.85E5	7.94E5
^{235}U	2.73E8	7.58E8	4.45E8	1.33E9	1.04E9	8.59E8
^{236}U	8.58E7	1.51E5	1.15E5	4.13E5	3.05E5	4.01E5
^{238}U	7.61E7	1.26E5	9.60E4	3.47E5	2.62E5	3.44E5
^{237}Np	5.55E8	1.08E8	5.93E7	2.31E8	1.30E8	1.33E8
^{239}Np	2.13E9	8.24E8	4.29E8	1.48E9	1.04E9	9.38E8
^{238}Pu	7.45E7	1.21E5	8.25E4	3.53E5	3.17E5	4.30E5
^{239}Pu	4.63E7	2.77E5	1.41E5	5.85E5	3.87E5	4.25E5
^{240}Pu	7.61E7	1.25E5	8.73E4	3.63E5	3.14E5	4.24E5
^{241}Pu	4.78E7	0.0	0.0	0.0	0.0	0.0
^{242}Pu	5.94E7	1.05E5	7.19E4	3.00E5	2.54E5	3.42E5
^{241}Am	1.59E8	7.64E7	5.49E7	1.90E8	8.98E7	1.04E8
^{243}Am	1.39E8	2.10E8	1.28E8	4.68E8	2.40E8	2.68E8
^{242}Cm	6.91E7	1.22E5	7.77E4	3.88E5	3.54E5	4.88E5
^{243}Cm	1.12E9	6.23E8	3.29E8	1.10E9	7.99E8	7.08E8
^{244}Cm	5.99E7	6.27E4	3.72E4	2.38E5	2.74E5	3.82E5

Table 7.3 External dose conversion factors for exposure to contaminated ground surface (millirem/year/μCi/Cm^2)

	Body surface (β dose rate)	Lungs	Ovaries	Skeleton	Testes	Total body
		(γ and x-ray dose rate)				
^{3}H	0.0	0.0	0.0	0.0	0.0	0.0
^{14}C	0.0	0.0	0.0	0.0	0.0	0.0
^{22}Na	2.22E5	2.20E6	1.40E6	2.64E6	2.21E6	2.33E6
^{24}Na	7.87E6	3.56E6	3.63E6	3.87E6	2.38E6	3.76E6
^{41}Ar	6.00E6	1.23E6	1.04E6	1.39E6	1.04E6	1.30E6
^{54}Mn	0.0	8.19E5	4.63E5	9.89E5	8.37E5	8.69E5
^{55}Fe	0.0	1.91E1	6.33E9	2.17E1	1.05E2	1.44E2
^{58}Co	8.52E3	9.70E5	5.16E5	1.19E6	1.03E6	1.03E6
^{60}Co	6.68E3	2.40E6	1.96E6	2.72E6	2.07E6	2.54E6
^{59}Ni	0.0	2.23E1	1.19E1	4.07E1	1.98E2	2.71E2
^{65}Zn	6.01E-4	5.56E5	3.96E5	6.44E5	5.12E5	5.88E5
^{85}Kr	1.26E6	2.36E3	9.42E2	3.05E3	2.83E3	2.53E3
^{88}Kr	2.84E6	1.76E6	1.76E6	1.95E6	1.18E6	1.84E6
^{88}Rb	1.95E7	5.96E5	5.76E5	6.56E5	4.07E5	6.24E5
^{89}Sr	8.57E6	1.32E2	7.92E1	1.57E2	1.31E2	1.40E2
^{90}Sr	2.74E5	0.0	0.0	0.0	0.0	0.0
^{90}Y	1.31E7	1.19E-1	6.73E-2	6.44E-1	7.92E-1	9.65E-1
^{91m}Y	4.41E5	5.54E5	2.35E5	7.09E5	6.46E5	5.93E5
^{91}Y	8.88E6	3.46E3	2.71E3	3.95E3	3.04E3	3.66E3
^{95}Zr	2.29E4	7.38E5	3.84E5	9.06E5	7.84E5	7.84E5
^{95}Nb	0.0	7.61E5	4.05E5	9.31E5	8.01E5	8.09E5
^{99}Tc	0.0	0.0	0.0	0.0	0.0	0.0
^{103}Ru	8.98E4	4.97E5	1.97E5	6.45E5	5.98E5	5.32E5
^{106}Ru	0.0	0.0	0.0	0.0	0.0	0.0
^{103m}Rh	0.0	2.32E2	2.22E2	1.41E3	1.94E3	1.36E3
^{106}Rh	1.70E7	2.11E5	9.87E4	2.66E5	2.39E5	2.25E5
^{110m}Ag	3.39E4	2.71E6	1.70E6	3.24E6	2.70E6	2.88E6
^{110}Ag	1.54E7	3.10E4	1.50E4	3.87E4	3.42E4	3.31E4
^{125}Sb	1.05E5	4.42E5	1.98E5	5.91E5	5.34E5	4.80E5
^{125m}Te	0.0	1.11E4	8.45E3	3.68E4	3.42E4	2.50E4
^{127m}Te	3.66E4	3.10E3	2.37E3	1.05E4	9.96E3	7.21E3
^{127}Te	8.24E5	5.25E3	2.30E3	7.28E3	6.62E3	5.69E3
^{129m}Te	3.05E6	3.20E4	1.67E4	4.48E4	3.97E4	3.71E4
^{129}Te	7.22E6	5.90E4	2.77E4	7.98E4	7.24E4	6.50E4
^{129}I	0.0	1.32E4	9.89E3	4.02E4	3.41E4	2.57E4
^{131}I	3.25E5	4.12E5	1.91E5	5.78E5	5.22E5	4.49E5
^{133m}Xe	0.0	3.58E4	2.19E4	7.05E4	6.23E4	4.77E4
^{133}Xe	1.56E0	4.01E4	2.45E4	9.70E4	5.79E4	5.57E4
^{135m}Xe	1.31E6	4.53E5	1.85E5	5.87E5	5.41E5	4.87E5
^{135}Xe	2.48E6	2.74E5	1.49E5	4.25E5	3.77E5	3.04E5
^{134}Cs	5.17E5	1.57E6	7.99E5	1.94E6	1.69E6	1.67E6
^{135}Cs	0.0	0.0	0.0	0.0	0.0	0.0
^{137}Cs	3.99E5	0.0	0.0	0.0	0.0	0.0
^{137m}Ba	1.19E6	6.06E5	2.91E5	7.59E5	6.70E5	6.47E5
^{144}Ce	0.0	2.04E4	1.16E4	4.19E4	2.65E4	2.50E4
^{144m}Pr	0.0	6.47E3	4.98E3	1.85E4	1.26E4	1.12E4
^{144}Pr	1.55E7	2.94E4	2.55E4	3.34E4	2.27E4	3.10E4

Table 7.3 (continued)

	Body surface (β dose rate)	Lungs	Ovaries	Skeleton	Testes	Total body
		(γ and x-ray dose rate)				
^{147}Pm	0.0	3.82E0	2.00E0	7.57E0	4.54E0	4.43E0
^{151}Sm	0.0	1.08E0	9.98E-1	6.14E0	8.50E0	6.24E0
^{154}Eu	1.77E6	1.20E6	8.15E5	1.47E6	1.15E6	1.28E6
^{155}Eu	0.0	6.19E4	3.50E4	1.38E5	7.22E4	7.73E4
^{207}Tl	6.69E6	2.09E3	1.24E3	2.49E3	2.08E3	2.21E3
^{208}Tl	8.10E6	2.94E6	2.72E6	3.32E6	2.13E6	3.11E6
^{209}Tl	9.62E6	2.08E6	1.74E6	2.49E6	1.82E6	2.22E6
^{209}Pb	4.80E5	0.0	0.0	0.0	0.0	0.0
^{210}Pb	0.0	1.66E3	1.32E3	4.42E3	2.62E3	3.05E3
^{211}Pb	5.90E6	5.14E4	2.59E4	6.60E4	5.77E4	5.51E4
^{212}Pb	2.01E4	1.61E5	9.02E4	2.75E5	2.16E5	1.83E5
^{214}Pb	9.16E5	2.68E5	1.32E5	4.02E5	3.45E5	2.96E5
^{210}Bi	4.51E6	0.0	0.0	0.0	0.0	0.0
^{211}Bi	6.53E2	4.90E4	2.30E4	7.16E4	6.32E4	5.37E4
^{212}Bi	6.37E6	1.80E5	1.23E5	2.14E5	1.71E5	1.92E5
^{213}Bi	5.36E6	1.42E5	6.14E4	1.93E5	1.74E5	1.53E5
^{214}Bi	8.50E6	1.44E6	1.17E6	1.65E6	1.20E6	1.51E6
^{210}Po	0.0	8.40E0	4.62E0	1.02E1	8.70E0	8.92E0
^{213}Po	0.0	3.01E1	16.2E1	3.68E1	3.15E1	3.20E1
^{214}Po	0.0	1.08E2	5.91E1	1.32E2	1.13E2	1.15E2
^{217}At	0.0	3.17E2	1.28E2	4.18E2	3.88E2	3.41E2
^{219}Rn	5.46E2	6.11E4	2.96E4	9.02E4	8.00E4	6.71E4
^{220}Rn	0.0	5.76E2	2.41E2	7.38E2	6.75E2	6.16E2
^{221}Fr	0.0	3.42E4	1.99E4	5.70E4	4.80E4	3.85E4
^{223}Fr	3.53E6	5.91E4	3.69E4	1.20E5	7.30E4	7.39E4
^{223}Ra	2.74E1	1.43E5	7.47E4	2.53E5	1.80E5	1.64E5
^{224}Ra	0.0	1.0E4	6.11E3	1.76E4	1.52E4	1.22E4
^{225}Ra	7.03E0	8.94E3	6.96E3	2.51E4	1.57E4	1.53E4
^{226}Ra	0.0	7.43E3	4.41E3	1.30E4	1.03E4	8.45E3
^{228}Ra	0.0	3.45E-5	1.84E-5	6.31E-5	3.07E-4	4.19E-4
^{225}Ac	0.0	1.70E4	9.17E3	3.34E4	2.11E4	2.04E4
^{227}Ac	0.0	1.31E2	6.75E1	2.69E2	1.70E2	1.77E2
^{228}Ac	4.15E6	9.04E5	5.90E5	1.11E6	9.00E5	9.64E5
^{227}Th	0.0	1.15E5	6.27E4	1.89E5	1.55E5	1.31E5
^{228}Th	0.0	2.17E3	1.23E3	4.32E3	2.98E3	2.95E3
^{229}Th	0.0	9.46E4	5.25E4	1.93E5	1.20E5	1.17E5
^{230}Th	0.0	4.38E2	2.70E2	9.77E2	7.69E2	8.92E2
^{231}Th	0.0	1.55E4	8.85E3	3.65E4	2.37E4	2.55E4
^{232}Th	0.0	2.09E2	1.31E2	5.11E2	4.77E2	6.10E2
^{234}Th	0.0	8.57E3	4.87E3	1.93E4	9.95E3	1.11E4
^{231}Pa	0.0	3.31E4	1.70E4	5.33E4	4.64E4	3.96E4
^{233}Pa	2.00E4	2.08E5	1.02E5	3.31E5	2.68E5	2.34E5
^{234m}Pa	1.17E7	1.09E4	6.70E3	1.34E4	1.08E4	1.16E4
^{234}Pa	1.70E6	1.96E6	1.19E6	2.47E6	2.00E6	2.10E6
^{232}U	0.0	3.40E2	2.04E2	8.33E2	8.24E2	1.05E3
^{233}U	0.0	3.58E2	1.81E2	7.86E2	6.18E2	7.33E2

Table 7.3 (continued)

	Body surface (β dose rate)	Lungs	Ovaries	Skeleton	Testes	Total body
			(γ and x-ray dose rate)			
^{234}U	0.0	1.77E2	1.07E2	4.82E2	5.53E2	7.43E2
^{235}U	0.0	1.67E5	9.80E4	2.93E5	2.31E5	1.91E5
^{236}U	0.0	9.96E1	6.17E1	3.12E2	4.43E2	6.21E2
^{238}U	0.0	8.59E1	5.32E1	2.71E2	3.89E2	5.46E2
^{237}Np	0.0	2.64E4	1.46E4	5.78E4	3.49E4	3.60E4
^{239}Np	3.58E4	1.84E5	9.58E4	3.33E5	2.33E5	2.13E5
^{238}Pu	0.0	1.08E2	5.90E1	3.88E2	5.59E2	7.97E2
^{239}Pu	0.0	9.11E1	4.56E1	2.44E2	2.71E2	3.60E2
^{240}Pu	0.0	1.06E2	5.91E1	3.78E2	5.36E2	7.64E2
^{241}Pu	0.0	0.0	0.0	0.0	0.0	0.0
^{242}Pu	0.0	8.47E1	4.69E1	3.00E2	4.24E2	6.03E2
^{241}Am	0.0	2.20E4	1.58E4	5.50E4	2.69E4	3.16E4
^{243}Am	0.0	5.23E4	3.21E4	1.22E5	6.05E4	6.79E4
^{242}Cm	0.0	1.21E2	6.29E1	4.94E2	6.28E2	9.05E2
^{243}Cm	0.0	1.39E5	7.33E4	2.48E5	1.80E5	1.62E5
^{244}Cm	0.0	9.80E1	4.91E1	4.21E2	5.59E2	8.07E2

It is important to emphasize that contributions from daughter nuclides, which may be present in a given source term or which will build up due to decay of the parent radionuclide, are not included in the parent nuclide's dose conversion factor. Care must be taken, when preparing a source term to be used in the calculation of dose from a fuel cycle facility, to include such daughter nuclides as will build up during the exposure period in question, either during atmospheric transport or following deposition. External dose conversion factors for all significant daughters of breeder cycle radionuclides of interest have also been included in Tables 7.2 and 7.3, to be used in such calculations. The reader is referred to Kocher[6] for a detailed discussion of this methodology, and for diagrams of half-lives and branching fractions of radionuclides of interest.

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8. DOSES DUE TO ATMOSPHERIC RELEASES OF ^{3}H AND ^{14}C

Elizabeth L. Etnier

8.1 Tritium (^{3}H)

8.1.1 Introduction

Various methodologies exist for the estimation of ingestion dose from continuous atmospheric releases of tritium.[1-7] Most of these methodologies are based on the specific activity model (according to which the tritium content of the plant or body water, related to the content of stable hydrogen, is the same as that of the water in air), but the resulting dose estimates may vary by up to a factor of 10 depending on the equilibrium assumptions employed for vegetation, beef, and milk.

Bush,[1] Evans,[2] Killough and McKay,[3] and Moore[4] base their dose calculations on the assumption that body hydrogen or body water is uniformly labeled with tritium (an equilibrium ratio of 1.0). However, their approaches are somewhat different in that Bush[1] and Evans[2] estimate dose based on the average energy delivered by HTO to body water that contains 1 μCi/liter. Killough and McKay,[3] and Moore,[4] follow the food and water ingestion pathway.

The U.S. Nuclear Regulatory Commission,[5] and Ng et al.[6] assume various uptake fractions for tritium in plants, beef, and milk, resulting in doses 1/4-1/2, respectively, those estimated by the Killough and McKay,[3] and Moore[4] methodologies.

Vogt[7] recommends modifying the basic specific activity model so as to consider the water supply of plants from the soil water and precipitation deposited on the leaves, as well as that from humidity.

A review of the literature available for estimating dose from atmospheric releases of tritium, as well as the range of reported values for the input parameters is being undertaken at the Oak Ridge National Laboratory (ORNL). Parameters being considered include: quality factor, half-life, air-to-water dilution, and organic binding of tritium.

At present, the methodology outlined in the AIRDOS-EPA computer code[8] is being employed at ORNL for estimating tritium doses. This methodology has been modified slightly from that reported by Moore[4] and may be modified further to reflect the changes suggested by the literature review in progress.

8.1.2 The AIRDOS-EPA methodology for ^{3}H

The AIRDOS-EPA code[8] treats tritium in the following manner: if tritium (T) is released to the atmosphere as HT or T_2, atoms of T may exchange with hydrogen atoms in water molecules in the air, and the plume is treated as though it contained HTO initially. The tritium may then be assumed to follow water almost precisely through the environment. For this reason, doses from drinking water are included for tritium. Rather than attempting to relate the doses to the ground deposition rate, it is assumed that doses from ingestion of food and drinking water at an environmental location are proportional to the tritium concentration in air.[3]

The total ingestion dose from tritium if the source of all of an individual's food and drinking water is assumed to be at his specific environmental location is:

$$D_t = C_f X + C_w X \quad , \tag{8.1}$$

where

D_t = tritium ingestion dose from food and water (rem/year),

C_f = dose conversion factor for food (rem-cm^3/pCi-year),

C_w = dose conversion factor for drinking water (rem-cm^3/pCi-year), and

χ = groundlevel concentration of tritium in air at the environmental location (pCi/cm^3).

The total-body dose conversion factor for ingestion is 8.3 x 10^{-5} rem/μCi.[10] This number is used to derive the value of C_f and C_w, based on the specific activity of tritium in atmospheric moisture with an average specific humidity of 8 g H$_2$O/m^3 of air (ref. 3). If tritium in food is in equilibrium with atmospheric tritium and man consumes 1638 g of water daily in his food, C_f is 6.18 rem-cm^3/pCi-year. The C_w value for an assumed daily drinking water intake of 1512 g is 5.70 rem-cm^3/pCi-year. This value is used, however, only if the source of each individual's drinking water is assumed to be at his specific environmental location. For all other cases, C_w is reduced to account for dilution by distant sources.

The code artificially breaks down the ingestion dose from tritium into percentage contributions of 50.5% from vegetables (including fruits and grains), 17.5% from meat and 31.0% from milk. The percentages are based on approximate water contents of foods: 82.4% for vegetables, 62.3% for meat, and 87.5% for milk, for daily intakes of 0.532 kg vegetables, 0.258 kg meat, and 0.307 kg milk (Sect. 6).

Tritium doses via inhalation of air and skin absorption are estimated by the code and added to the dose estimated to be received via ingestion. The dose conversion factor for inhalation[10] of air containing tritium includes a 50% contribution for skin absorption from air.

8.2 Carbon-14 (^{14}C)

8.2.1 Introduction

Methods for calculating dose to man from atmospheric releases of ^{14}C are also being reviewed. Basically, these methods involve a specific activity approach, although they differ in their treatment of pathways. The NRC (ref. 5) assumes that the ratio in vegetation of ^{14}C to natural carbon is the same as the ratio in the atmosphere of ^{14}C to natural carbon. The ingestion of ^{14}C in foods, including various transfer coefficients for vegetation-to-beef and vegetation-to-milk, is then used to calculate the ultimate dose to man. For intermittent releases, an account is made for achievement of a fractional equilibrium ratio by relating total annual releases time to the total annual time during which photosynthesis occurs.

The methodology currently in use at ORNL, and described in the AIRDOS-EPA code,[8] assumes that the ^{14}C in body tissues is in equilibrium with the ^{14}C in the atmosphere during continuous releases. Since about 99% of the dose from ^{14}C is via the ingestion pathway,[9] a breakdown based on percent contribution to diet of the three major food pathways, as well as the carbon content of each food, is utilized in the AIRDOS-EPA code to facilitate a pathway analysis.

8.2.2 The AIRDOS-EPA methodology for ^{14}C

The description of the treatment of ^{14}C in the AIRDOS-EPA code[8] is as follows: if ^{14}C is released in the form of $^{14}CO_2$, it will mix with atmospheric CO_2 and become available for plant photosynthesis. Cattle grazing on pasture will take in ^{14}C in grass, and then man will receive

it in milk and beef. Factors in a data statement are used in the code to multiply by the concentration of ^{14}C in air to obtain an ingestion dose for each reference organ. These dose conversion factors, listed in Table 8.1 are based on specific activity calculations for ^{14}C in body tissues in equilibrium with ^{14}C in the atmosphere.[9]

Nearly all the ^{14}C dose comes from ingestion. An artificial breakdown by the three food pathways is accomplished by estimating carbon intakes for vegetables, milk, and meat, based on a carbon content of approximately 8% for vegetables, 24% for meat, and 7% for milk.

Table 8.1. Dose conversion factors for ^{14}C

Organ	Dose conversion factors[a] (rem-cm^3/pCi-year)
Whole body	1.16×10^3
Red marrow	2.03×10^3
Lungs	5.07×10^2
Endosteal cells	1.85×10^3
Stomach wall	7.43×10^2
Lower large intestine wall	8.92×10^2
Thyroid	5.27×10^2
Liver	7.30×10^2
Kidneys	6.49×10^2
Testes	4.46×10^2
Ovaries	4.46×10^2

[a]These factors are taken from ref. 8 and are based on the assumption that the specific activity in human tissue is equal to the average steady state value in the atmosphere (ref. 9).

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Appendix A

A COMPARISON OF EXPOSURE PATHWAYS

Craig A. Little

Releases of radionuclides to the environment from nuclear facilities, including breeder reactors, are generally either airborne or waterborne. In either case, there are several pathways by which humans may be exposed to radioactivity from either type of release. The main body of this report presents information on the models and parameter values which may be used to estimate the dose to humans as a result of these exposures. The purpose of this appendix is to illustrate for both release types the relative contribution of various pathways of exposure to the total dose due to radionuclide releases from various nuclear facilities.

This comparison is accomplished through a review of pathway comparisons performed for various facilities of the light water reactor (LWR) fuel cycle.

The primary reference for the discussion of atmospheric releases was published by Hoffman and Kaye.[1] Most of the data mentioned here were taken from tables published in that paper. A primary reference for the importance of various pathways following an aquatic release is a paper by Soldat[2] from the same symposium volume. While these examples do not include breeder reactor facilities, the models used in these comparisons are basically the same as those discussed in this report. As such, they should serve the illustrative purposes desired for this appendix. Further, no single comparison or sets of comparison can define the critical pathway or pathways to man from a given type of facility since any such comparison depends on the radionuclide make-up of the release, the rate of release, the dietary and behavioral habits

of the human receptor, and the existence of a particular pathway at a
given site.

A.1 Atmospheric Release

Hoffman and Kaye[1] calculated the maximum individual exposure from
the major terrestrial pathways for atmospheric releases from model LWR
fuel cycle facilities. Each model facility resulted in a dose to the
organs of some limiting individual (adult, child or both) after assuming
a generic location and respiratory parameters designed to maximize the
exposure to humans via that pathway. These doses were then compared to
applicable radiation standards to assess the significance of the particu-
lar pathway. For each facility, the principal nuclides and principal
exposure pathways for organs of the limiting child or adult were listed.
Rather than reproduce those seven tables here, we have summarized the
information in Table A.1. A reader desiring the detail of the original
report should see Tables 8-14 of ref. 1.

There are several observations about the pathways which may be made
from the information in Table A.1. One conclusion is that for the con-
sidered release and human behavioral conditions milk is an extremely
important pathway of exposure, especially for children. Of the 31
entries in the rightmost column in Table A.1, "principal pathways for
principal nuclides," 16 are milk ingestion. Milk ingestion was often
the most important pathway of radiation exposure when children alone
were considered.

For adults, milk ingestion was slightly less important, but milk,
vegetable, and beef ingestion still accounted for one-half of the prin-
cipal pathway entries. Immersion in contaminated air was an important

Table A.1 Principal pathways and nuclides for maximally exposed individuals
from model facilities in the uranium fuel cycle[a]

Source	Limiting receptor	Principal nuclides	Reference organ	Principal pathways for principal nuclides[b]
Uranium mill	Child	^{226}Ra	Bone, lung, thyroid, total body	Milk (93-94%)
	Adult	^{226}Ra	GI, liver	Milk (58-59%)
		^{238}U	GI	Vegetables (58-80%)
		^{210}Po	Kidney, liver	Vegetables (67%)
		^{230}Th	Kidney, liver	Inhalation (97%)
		^{210}Pb	Liver	Vegetable (80%)
Conversion facility	Child	^{226}Ra	Lung, thyroid, total body	Milk (93%)
		^{234}U	Lung	Inhalation (93%)
	Adult	^{230}Th	Bone, kidney	Inhalation (97-98%)
		^{226}Ra	Bone	Milk (60%)
Uranium enrichment facility	Child	^{234}U	Bone	Milk (56%)
		^{234}U	Lung	Inhalation (92%)
Fuel fabrication plant	Child	^{234}U	Bone	Milk (56%)
Boiling water reactor	Child	^{131}I	Thyroid	Milk (99%)
		^{88}Kr	Thyroid	Immersion in air (100%)
	Adult	^{88}Rb	GI	Inhalation (72%)
		^{88}Kr	GI	Immersion in air (100%)
	Child/Adult	^{88}Kr	Bone, skin, total body, lung, liver, kidney	Immersion in air (100%)
		^{14}C	Bone, total body	Milk (88%)
		^{133}Xe	Skin	Immersion in air (100%)
		^{88}Rb	Lung, liver, kidney	Immersion in air (59-76%)
Fuel reprocessing	Child	^{131}I	Thyroid	Milk (96%)
		^{129}I	Thyroid, total body, kidney	Milk (59-75%)
		^{90}Sr	Bone, lung	Milk (83-93%)
		^{3}H	Bone, liver, total body, kidney, lung	Milk (54-96%)
		^{134}Cs	Liver	Milk (84%)
		^{14}C	Bone, total body	Milk (88%)
	Adult	^{129}I	Thyroid	Milk (46%)
		^{131}I	Thyroid	Milk (77%)
		^{106}Ru	GI	Beef (99%)
	Child/Adult	^{85}Kr	Skin	Immersion in air (100%)

[a]Data taken from Tables 8-14 of ref. 1.

[b]Range of percentages indicates the contribution by the principal nuclide to the organs
of reference over the listed pathway.

pathway for the boiling water reactor and the fuel reprocessing plant. Inhalation was a principal pathway for adults subject to releases from uranium mill and the uranium conversion facilities and for children near uranium enrichment facilities. Exposure to contaminated surfaces was found to be relatively unimportant for the releases considered in this paper.

As might be expected, the principal radionuclides released were not the same for each facility type. Important releases early in the fuel cycle tend to be composed of naturally occurring radionuclides, while the principal radionuclides released by the reactor and reprocessing plant are more frequently fission products.

It is apparent from this brief discussion that the importance of the various pathways of exposure is very much a function of the nuclides released. Therefore, it follows that the importance of any pathway varies between types of facilities and within types of facilities, depending upon the particular radionuclides present and the emission control equipment installed.

Studies similar to the one of Hoffman and Kaye[1] were performed by various authors, but for specific types of facilities and for population dose commitments rather than maximum individual doses (Table A.2). The data in Table A.2 from these studies are not identified by radionuclide, but the conclusions drawn from Table A.1 hold for these other studies; namely, ingestion and inhalation appear to be the most important pathways, but the relative rankings of the pathways varies between and within facility types due to differences in assumptions about radionuclide releases and environmental transport factors.

Table A.2 Fraction of the 50-year dose commitment to the population within 50 miles of each fuel cycle facility from gaseous releases per year of operation of the facility

| Facility | Fraction of 50 mile population dose commitment | | | | |
	Immersion	Contaminated ground surface	Inhalation	Ingestion	References
Mill[a]	<1.0E-3	2.0E-3	4.9E-2	9.5E-1	3
Mill[a]	<1.0E-3	1.1E-1	1.5E-1	7.4E-1	4
Fuel fabrication UO$_2$	<1.0E-3	3.2E-1	1.9E-1	4.9E-1	3
Mixed oxide	<1.0E-3	1.0E-2	9.8E-1	8.0E-3	3
Fuel fabrication	<1.0E-3	7.4E-1	2.1E-1	4.4E-2	4
Reprocessing[a]	2.2E-1	1.2E-1	5.7E-1	9.9E-2	3
Reprocessing[a]	3.7E-2	2.2E-1	1.3E-1	6.2E-1	4

[a]Both references 3 and 4 considered both a mill and a reprocessing plant. The results differ because of differences in source term and assumptions about the facility.

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A.2 Aquatic Releases

We are aware of no paper examining the importance of various aquatic pathways which is directly comparable to the paper by Hoffman and Kaye.[1] An adjoining paper by Soldat[2] in the same volume discusses "only the potential aquatic exposure pathways, including drinking water, aquatic foods, swimming and boating, contaminated sediments, and crops and animal products from irrigated farms." Soldat[2] goes on to say that the importance of the various pathways cannot be evaluated without knowing the release rates and physicochemical form of the radionuclides, the type of water receiving effluent and its use by humans.

Given these precautions, Soldat[2] discusses several studies of population doses from aquatic releases and attempts to compare the importance of the various pathways. Results from the studies cited by Soldat[2] and two additional studies are summarized in Table A.3. Different assumptions were used in each of these so care must be exercised when comparing them.

However, these four studies do reaffirm the point made in the atmospheric section that the importance of the pathway varies greatly depending on the organ of concern, the type of release, and the assumptions made when calculating doses, as well as the relationship between the calculated dose and a dose limit. The only apparent exception to this rule is the swimming pathway which, for the studies referenced, never contributed more than 3% of the total calculated dose.

Aside from the constant small contribution of the swimming pathway, the relative importance of the various pathways changes frequently due to

Table A.3 Fraction of the calculated total dose from one or more radionuclides listed by aquatic pathway in several studies

| Organ | Radionuclide | Source of release | Fraction of total dose contribution from radionuclide given by | | | | Reference |
			Drinking water	Fish ingestion	Irrigation	Swimming	
Total body	Mixed	LWR	0.54	0.46	NC[a]	NC[a]	b
	Mixed	LWR	0.27	0.54	0.19	NC	c
	Mixed	LMFBR	0.89	0.03	0.06	0.03	4
	^{3}H	Waste burial	0.71	0.01	0.27	d	8
	^{90}Sr	Waste burial	0.07		0.93		8
	^{106}Ru	Waste burial	0.62	0.11	0.27		8
	^{137}Cs	Waste burial	0.11	0.75	0.14		8
Thyroid	Mixed	LWR	0.97	0.03	NC	NC	b
	Mixed	LWR	0.63	0.13	0.25	NC	c
	^{3}H	Waste burial	0.71	0.01	0.27		8
	^{90}Sr	Waste burial	0.92	0.08			8
	^{106}Ru	Waste burial	0.22	0.04	0.75		8
	^{137}Cs	Waste burial	0.13	0.87			8
Bone	Mixed	LWR	0.52	0.48	NC	NC	b
	Mixed	LWR	0.08	0.71	0.21	NC	c
	Mixed	LMFBR	0.97		0.03		4
	^{3}H	Waste burial	0.71	0.01	0.27		8
	^{90}Sr	Waste burial	0.45	0.04	0.52		8
	^{106}Ru	Waste burial	0.62	0.10	0.28		8
	^{137}Cs	Waste burial	0.11	0.80	0.09		8

[a]No calculation (NC) pathway not considered.
[b]Robbins and Martin[6] as cited by Soldat.[2]
[c]Schuckler, Kalckbrenner, and Bayer[7] as cited by Soldat.[2]
[d]Contribution of this pathway is less than 1% of the total

changes in the assumptions used in the calculations. The drinking water pathway contributes from 8 to 97% of the total dose. The fish ingestion pathway varies from negligible (less than 1%) to as much as 87%. Finally, the irrigation pathway contributed from less than 1% to as much as 93%. It is important to remember that the range of contributions of each pathway may not be completely specified in Table A.3; site specific considerations for almost any parameter or variable in the dose equations may significantly alter the contribution to dose of any given parameter.

Likely the most reasonable conclusion of this short review is that no single aquatic pathway to man dominates the dose calculations. Rather, the only pathway which is not very important, primarily as a result of low exposure possibility, is the swimming pathway. The only time this pathway would likely become important is if the other pathways were nonexistent at the site being considered.

A.3 Relative Contribution of Atmospheric vs Aquatic Releases

In general, an examination of the references cited in this appendix as well as other reference sources indicates that, for both individual and population doses, atmospheric releases from nuclear facilities are expected to contribute a larger fraction of the dose to humans than are aquatic releases. One reason for this expectation is that the source term for atmospheric releases is generally higher than the source term for aquatic releases. Also, humans tend to have more interaction with airborne than with waterborne radionuclides. However, variations between sites can be large enough that conditions prevailing at a specific site

should be considered when attempting to determine the relative importance of aquatic or atmospheric exposures in the calculation of a specific maximally exposed individual or population dose.

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Appendix B

EXAMPLE CALCULATIONS

Charles W. Miller

The body of this report presents models and parameter values which may be used to assess the impact on man of radionuclides released to the environment by breeder reactor and other nuclear facilities. The purpose of this appendix is to illustrate how these models and parameter values are used in numerical calculations. Example problems are solved for environmental concentrations and subsequent doses to humans as a result of releases to the atmosphere and surface waters for a limited number of nuclides. While the examples considered are far from exhaustive when compared to those that might be encountered in a real assessment situation, it is hoped that these cases will be of assistance to persons attempting such calculations for the first time or to those who simply need to understand how such calculations are made. Also, these problems could be used as a partial check of computer implementations of material contained in this report.

When solving these example problems, reference is made to the location in the main body of the report where the equation, methodology, or parameter value being used is originally presented. The reader should consult the referenced locations for a complete definition of all of the terms used in solving the sample problem.

In the main body of this report, information is presented on the statistical distribution associated with the parameters whenever information on the distribution could be obtained. In this appendix, mean values of these parameters are used in the example calculations. It must be emphasized that the selection of the mean value was an arbitrary one for illustrative purposes only, and such use here does

not constitute a recommendation that mean values be used for all assessment calculations. In addition, the release rates used in these example problems were also chosen arbitrarily, and no relationship between them and actual release rates from breeder reactors or any other nuclear facilities was intended.

B.1 Environmental Concentrations

B.1.1 Releases to the Atmosphere

B.1.1.1 Air concentration from a ground-level release. Assume an annual release of 1×10^{-2} Ci of $^{131}I_2$ from a ground level (H = 0 m) source under Pasquill atmospheric stability category D. Because of the release height, the Pasquill-Gifford values of σ_y and σ_z should be used in the Gaussian plume model, Eq. (2.1). From Table 2.1, for a downwind distance of x = 2000 m,

$$\sigma_y = (a_1 \ln x + a_2)x \qquad (B.1)$$
$$= (-0.0059[\ln 2000] + 0.11)(2000)$$
$$= 130 \text{ m},$$

and

$$\sigma_z = \frac{1}{2.15} \exp(b_1 + b_2 \ln x + b_3 \ln^2 x) \qquad (B.2)$$
$$= \frac{1}{2.15} \exp(-1.35 + 0.793[\ln 2000] + 0.0022[\ln 2000]^2)$$
$$= 57 \text{ m} .$$

Equation (2.8) for plume depletion cannot be solved analytically. However, values of the ratio $\frac{Q'}{Q}$, the plume depletion fraction, for v_d = 1 cm/s and u = 1 m/s can be obtained from Fig. 5.5 of Meteorology

and Atomic Energy – 1968 (ref. 1). For a downwind distance of 2000 m, a release height of 0 m, and Pasquill D stability, Fig. 5.5 gives $\frac{Q'}{Q}$ = 0.35. Assume that the actual wind speed for the $^{131}I_2$ release being considered here is 3 m/s. Further assume that v_d(total) = 3.5 cm/s (Sect. 2.2.1) and v_w = 1.3 cm/s (ref. 2). This results in a total deposition velocity of 4.8 cm/s. The value of $\frac{Q'}{Q}$ noted above can now be changed to reflect the assumed release conditions by

$$\left(\frac{Q'}{Q}\right)_2 = \left(\frac{Q'}{Q}\right)_1^{\frac{u_1 v_2}{u_2 v_1}} \tag{B.3}$$

$$= (0.35)^{\frac{(1 \text{ m/s})(4.8 \text{ cm/s})}{(3 \text{ m/s})(1 \text{ cm/s})}}$$

$$= 0.19 \ .$$

This indicates that the deposition processes considered here result in an air concentration at 2000 m that is 19% of the air concentration that would be expected if these deposition processes were not considered.

The half-life, $t_{1/2}$, of ^{131}I is 8.05 days (ref. 3). Therefore, the radiological decay constant, λ, is 9.96 x 10^{-7} s^{-1}.

Equation (2.1) may be used to calculate the ground-level, centerline air concentration. Accounting for decay and plume depletion, Eq. (2.1) becomes

$$\chi = \left[\frac{Q}{\pi u \sigma_y \sigma_z} \ \exp\left(-\lambda\frac{x}{u}\right)\right] \left(\frac{Q'}{Q}\right)_2 \tag{B.4}$$

$$Q = 1 \times 10^{-2} \text{ Ci/year} = 3.2 \times 10^{-10} \text{ Ci/s}$$

$$\chi = \left[\frac{(3.2 \times 10^{-10} \frac{\text{Ci}}{\text{s}})}{\pi(3\text{m/s})(130 \text{ m})(57 \text{ m})} \exp \left(-[9.96\text{E}-7][2000/3]\right) \right] (0.19)$$

$$= 8.7 \times 10^{-16} \text{ Ci/m}^3$$

The 22.5° sector average air concentration, based on Eq. (2.2), is given by

$$\chi = \left[\frac{2.032 \, Q}{x u \sigma_z} \exp(-\lambda\frac{x}{u}) \right] \left(\frac{Q'}{Q}\right)_2 \tag{B.5}$$

$$= \left[\frac{(2.032)(3.2 \times 10^{-10} \frac{\text{Ci}}{\text{s}})}{(2000 \text{ m})(3 \text{ m/s})(57 \text{ m})} \exp \left(-[9.96\text{E}-7][2000/3]\right) \right] (0.19)$$

$$= 3.6 \times 10^{-16} \frac{\text{Ci}}{\text{m}^3}$$

B.1.1.2 **Air concentration from an elevated release.** Assume an annual release of 100 Ci of ^{85}Kr from a 100-m tall stack. Since Kr is a nonreactive gas and it has a half-life of 10.76 years (ref. 3), radiological decay and plume depletion need not be considered in this calculation. Using the Brookhaven dispersion parameters for this elevated release and assuming class B_1 stability and a downwind distance of 2 km, Table 2.2 gives

$$\sigma_z = p_z x^{q_z} \tag{B.6}$$
$$= (0.33)[(2000 \text{ m})^{0.86}]$$
$$= 230 \text{ m}$$

If u = 4 m/s, the 22.5° sector-average air concentration for this release is, from Eq. (2.2),

$$\chi = \frac{2.032Q}{xu\sigma_z} \exp\left\{-1/2\left(\frac{H}{\sigma_z}\right)^2\right\} \tag{B.7}$$

$$= \frac{(2.032)(3.2 \times 10^{-6}\frac{Ci}{s})}{(2000\ m)(4\ m/s)(230\ m)} \exp\left\{-1/2\left(\frac{100\ m}{230\ m}\right)^2\right\}$$

$$= 3.2 \times 10^{-12}\ Ci/m^3$$

B.1.1.3 Ground deposition. The rate of dry deposition onto the earth's surface is given by Eq. (2.4) as

$$d = \chi v_d \tag{B.8}$$

Using the information contained in Sect. B.1.1.1 for a ground-level atmospheric release of $^{131}I_2$,

$$v_d = 3.5\ cm/s$$
$$\chi = 3.6 \times 10^{-16}\ Ci/m^3$$

and

$$d = \chi v_d$$
$$= (3.6 \times 10^{-16}\ Ci/m^3)(3.5 \times 10^{-2}\ m/s)$$
$$= 1.3 \times 10^{-17}\ Ci/m^2\text{-}s$$
$$= 4.7 \times 10^{-2}\ pCi/m^2\text{-}h$$

B.1.1.3.1 Concentration on grass. The concentration in and on grass as a result of the deposition rate calculated above is given by Eq. (3.1) as

$$C_i^v = d_i \left\{ \frac{R[1 - \exp(-\lambda_{Ei} t_e)]}{Y_v \lambda_{Ei}} + \frac{B_{iv}[1 - \exp(-\lambda_i t_b)]}{P\lambda_i} \right\} \exp(-\lambda_i t_h) \quad (B.9)$$

The input parameter values for this case are

d_i = 4.7 x 10^{-2} pCi/m^2-h (from above),

$\frac{R}{Y_v}$ = 2 m^2/kg (Table 3.1),

λ_{Ei} = 6.4 x 10^{-3} h^{-1} (ref. 4) ,

t_e = 720 h (Table 3.7),

B_{iv} = 2.0 x 10^{-1} (Table 3.2),

λ_i = 3.6 x 10^{-3} h^{-1} (ref. 3),

P = 215 kg/m^2 (Table 3.6),

t_b = 8.76 x 10^3 h (assumed),

t_h = 0 (Table 3.7) .

The resulting concentration of ^{131}I in or on the grass is

C_i^v = 15 pCi/kg .

B.1.1.3.2 <u>Concentration on fresh produce</u>. Equation (B.9) can also be used to estimate the concentrations in and on fresh produce as a result of the deposition of ^{131}I considered above. Assuming this fresh produce is leafy vegetables, the input parameter values for this case are

d_i = 4.7 x 10^{-2} pCi/m^2-h (from above),

R = 0.2 (Table 3.1),

Y_v = 1.9 kg/m^2 (Table 3.1),

λ_{Ei} = 6.4 x 10^{-3} h^{-1} (ref. 4),

t_e = 1440 h (Table 3.7),

B_{iv} = 5.5 x 10^{-2} (Table 3.3),

λ_i = 3.6 x 10^{-3} h^{-1} (ref. 3),

P = 215 kg/m^2 (Table 3.6),

t_b = 8.76 x 10^3 h (assumed),

t_h = 336 h (Table 3.7).

Entering these values into Eq. (B.9) results in a concentration of ^{131}I in or on leafy vegetables of

$$C_i^V = 2.3 \times 10^{-1} \text{ pCi/kg} \quad .$$

B.1.1.3.3 _Concentration in milk_. Assume that both beef and dairy cattle are getting 100% of their food from the grass considered in Sect. B.1.1.3.1. The resulting concentration of ^{131}I in milk produced by these dairy cattle is, from Eq. (3.2),

$$C_i^M = F_m C_i^V Q_F \exp(-\lambda_i t_f) \tag{B.10}$$

The input parameter values are

F_m = 1.0 x 10^{-2} day/liter (Table 3.4),

C_i^V = 15 pCi/kg (see above),

Q_F = 15.6 kg/day (Table 3.6),

t_f = 4 days (Table 3.7),

λ_i = 8.6 x 10^{-2} day^{-1} (ref. 3),

resulting in

C_i^M = 1.6 pCi/liter.

B.1.1.3.4 Concentration in beef. Similarly, the concentration of ^{131}I in beef may be found using Eq. (3.4),

$$c_i^F = F_f c_i^V Q_F \exp(-\lambda_i t_s)$$ (B.11)

and the new input values of

$$F_f = 7 \times 10^{-3} \text{ day/kg (Table 3.5)},$$

and

$$t_s = 20 \text{ days (Table 3.7)}.$$

This calculation results in

$$c_i^F = 2.9 \times 10^{-1} \text{ pCi/kg} .$$

B.1.2 Release to Surface Water

B.1.2.1 Surface water concentrations. The fundamental form of the model for surface water concentration without sorption is given by Eq. (4.1). This model cannot be solved in general, however, without resorting to computerized numerical techniques. If, however, we assume a vertical line source emitting a constant W Ci/s, a closed form solution of Eq. (4.2) can be obtained (ref. 5). If we further assume a uniform, straight, rectangular channel in which the water flows, the solution to Eq. (4.2) is given by (ref. 5)

$$C_i = \frac{W}{Q} \left[1 + 2 \sum_{n=1}^{\infty} \exp\left(- \frac{n^2\pi^2 x k_y}{uy^2} \right) \left\{ \cos^2\left(\frac{n\pi b}{y} \right) \right\} \right]$$ (B.12)

where

Q = total river flow (m^3/s),

b = cross-stream location of sampling point (m), and

the other parameters are as defined for Eq. (4.2). Using flow parameters based on Hudson River data (ref. 6) and a discharge rate of 1 Ci/s of ^{137}Cs, the input parameter values are

$W = 1 \times 10^{-6}$ Ci/s,

$Q = 1600$ m^3/s,

$K_y = 5$ m^2/s,

$u = 0.6$ m/s,

$y = 600$ m,

$x = 2000$ m,

$b = 200$ m .

Considering only the first three terms of the series expansion,

$$C_i = \frac{(1 \times 10^{-6} \text{Ci/s})}{(1600 \text{ m}^3\text{/s})} \left[1 + 2 \sum_{n=1}^{3} \exp\left(- \frac{n^2 \pi^2 [2000 \text{ m}][5\frac{m}{s}]^2}{[0.6 \text{ m/s}][600 \text{ m}]^2} \right) \right.$$

$$\left. \cdot \left\{ \cos^2 \left(\frac{n\pi[200 \text{ m}]}{[600 \text{ m}]} \right) \right\} \right]$$

$= (6.2 \times 10^{-10})[1 + 2(0.16 + 0.04 + 0.02)]$

$= 9 \times 10^{-10}$ Ci/m^3

$= 9 \times 10^{-13}$ Ci/liter

B.1.2.2 <u>Concentration in fish</u>. Assume that freshwater finfish inhabit the stream considered above. The concentration CF_i (Ci/kg) of ^{137}Cs in these fish is given by

$$CF_i = B_{ip}C_i \quad . \tag{B.13}$$

If we further assume that no site-specific information is available on the concentration of K in this stream, a mean value of B_{ip} can be obtained from Table 5.1. This results in

$$CF_i = (1900 \text{ liter/kg})(9.1 \times 10^{-13} \text{ Ci/liter})$$
$$= 1.7 \times 10^{-9} \text{ Ci/kg} \quad .$$

B.2 Dose Calculations

B.2.1 <u>External Doses</u>

B.2.1.1 <u>Immersion in contaminated air</u>. The external dose rate to an individual immersed in contaminated air is given by (ref. 7)

$$R_{ij}^{ea} = D_{ij} \chi_i \quad , \tag{B.14}$$

where

R_{ij}^{ea} = dose rate to organ j due to immersion in air contaminated with radionuclide i (millirem/year),

D_{ij}^{ea} = dose conversion factor due to immersion in air for radionuclide i and organ j (millirem/year per $\mu Ci/cm^3$),

χ_i^a = concentration of radionuclide i in air ($\mu Ci/cm^3$).

From Sect. B.1.1.1 above, a 22.5° sector-averaged air concentration for a ground-level release of $^{131}I_2$ was found to be

$$\chi_i^a = 3.6 \times 10^{-16} \text{ Ci/m}^3$$
$$= 3.6 \times 10^{-16} \text{ } \mu\text{Ci/cm}^3 \text{ .}$$

From Table 7.2 the dose conversion factor for the β dose rate to the body surface from $^{131}I_2$ is found to be

$$D_{ij}^{ea} = 1.71 \times 10^9 \text{ millirem/year per } \mu\text{Ci/cm}^3 \text{ .}$$

Therefore, for this release the annual β dose rate to the body surface is

$$R_{ij}^{ea} = (1.71 \times 10^9 \text{ } \frac{\text{millirem}}{\text{year}} \text{ } \frac{\text{cm}^3}{\mu\text{Ci}})(3.6 \times 10^{-16} \text{ } \frac{\mu\text{Ci}}{\text{cm}^3})$$
$$= 6.2 \times 10^{-7} \text{ millirem/year .}$$

Similar calculations can be performed for other organs listed in Table 7.2. The resulting γ and X-ray dose rates for all organs considered are as follows:

Organ	Dose rate, millirem/year
Body surface	6.2×10^{-7}
Lungs	7×10^{-7}
Ovaries	3.3×10^{-7}
Skeleton	9.8×10^{-7}
Testes	8.9×10^{-7}
Total body	7.6×10^{-7}

Section B.1.1.2 gives an air concentration for an elevated release of ^{85}Kr,

$$\chi_i^a = 3.2 \times 10^{-12} \text{ Ci/m}^3$$

$$= 3.2 \times 10^{-12} \text{ }\mu\text{Ci/cm}^3 \text{ .}$$

Using Eq. (B.14) and Table 7.2, the annual β and photon dose rates for this release condition are found to be the following:

Organ	Dose rate, millirem/year
Body surface	7.2×10^{-3}
Lungs	3.7×10^{-5}
Ovaries	1.5×10^{-5}
Skeleton	4.8×10^{-5}
Testes	4.4×10^{-5}
Total body	3.9×10^{-5}

B.2.1.2 <u>Exposure to contaminated ground</u>. The external dose rate to an individual as a result of exposure to a radionuclide deposited on the ground is found from (ref. 7)

$$R_{ij}^{cg} = d_i \frac{1 - \exp(-\lambda_T t)}{\lambda_T} D_{ij}^{cg} (8.64 \times 10^4) , \tag{B.15}$$

where

R_{ij}^{cg} = dose rate to organ j due to exposure to ground contaminated with radionuclide i (millirem/year),

d_i = rate of deposition onto ground of radionuclide i (μCi/cm$^2 \cdot$s),

λ_T = radioactive decay constant λ_r + environmental decay constant λ_w (day^{-1}),

t = time allotted for surface buildup (days),

D_{ij}^{cg} = dose conversion factor for radionuclide i and organ j due to surface exposure to an infinite plane at a point 1 m above ground (millirem/year per $\mu Ci/cm^2$),

8.64×10^4 = s/day.

Section B.1.1.3 gives a dry deposition rate onto the earth's surface resulting from a ground-level release of $^{131}I_2$,

$$d_i = 1.3 \times 10^{-17} \ Ci/m^2\text{-}s$$
$$= 1.3 \times 10^{-15} \ \mu Ci/cm^2\text{-}s \ .$$

If a one year surface buildup time and no losses from the surface due to weathering effects are assumed, then

$$\lambda_T = \lambda_r = 8.61 \times 10^{-2} \ d^{-1} \ ,$$
$$t = 365 \ d$$

and

$$R_{ij}^{cg} = (1.3 \times 10^{-15} \ \frac{\mu Ci}{cm^2 \cdot s}) \ \frac{(1 - \exp[-\{8.61 \times 10^{-2}d^{-1}\}\{365 \ d\}])}{(8.61 \times 10^{-2} \ d^{-1})}$$

$$\cdot \ (8.64 \times 10^4 \ \frac{s}{d}) \ D_{ij}^{cg}$$

$$= (1.3 \times 10^{-9} \ \frac{\mu Ci}{cm^2}) \ D_{ij}^{cg}$$

Values of D_{ij}^{cg} for various organs may be found in Table 7.3 For the β dose rate to the body surface from ^{131}I,

$$D_{ij}^{cg} = 3.25 \times 10^5 \ \frac{millirem}{year} \cdot \frac{cm^2}{\mu Ci} \ .$$

Therefore, for this tissue,

$$R_{ij}^{cg} = (1.3 \times 10^{-9} \frac{\mu Ci}{cm^2})(3.25 \times 10^5 \frac{millirem}{year} \cdot \frac{cm^2}{\mu Ci})$$

$$= 4.2 \times 10^{-4} \text{ millirem/year.}$$

Similar calculations for the remaining organs listed in Table 7.3 complete the following table of dose rates:

Organ	Dose rate, millirem/year
Body surface	4.2×10^{-4}
Lungs	5.4×10^{-4}
Ovaries	2.5×10^{-4}
Skeleton	7.5×10^{-4}
Testes	6.8×10^{-4}
Total body	5.8×10^{-4}

B.2.2 Internal Doses

B.2.2.1 Inhalation dose. The dose rate to various organs of the body as a result of inhaling contaminated air is given by (ref. 7)

$$R_{ij}^{br} = \chi_i^a \, B_r \, D_{ij}^{br} \quad , \tag{B.16}$$

where

R_{ij}^{br} = dose rate to organ j due to breathing air contaminated with radionuclide i (rem/year),

χ_i^a = concentration of radionuclide i in air $(\frac{\mu Ci}{m^3})$,

B_r = breathing rate $(\frac{m^3}{year})$,

D_{ij}^{br} = dose conversion factor for radionuclide i and organ j due to breathing of contaminated air (rem/μCi intake).

From Table 6.1,

$$B_r = 8.03 \times 10^3 \ m^3/year.$$

From Sect. B.1.1.1 for $^{131}I_2$,

$$\chi_i^a = 3.6 \times 10^{-16} \ Ci/m^3$$

$$= 3.6 \times 10^{-10} \ \mu Ci/m^3 \ .$$

From Table 7.1 for the thyroid,

$$D_{ij}^{br} = 1.13 \ \frac{rem}{\mu Ci} \ .$$

The resulting annual dose rate to the thyroid for this release of $^{131}I_2$ is then

$$R_{ij}^{br} = (3.6 \times 10^{-10} \ \frac{\mu Ci}{m^3})(8.03 \times 10^3 \ \frac{m^3}{year})(1.13 \ \frac{rem}{\mu Ci})$$

$$= 3.3 \times 10^{-6} \ rem/year$$

$$= 3.3 \times 10^{-3} \ millirem/year.$$

The doses to the remaining organs listed in Table 7.1 are also estimated from the application of Eq. (B.16) and included in the following:

Organ	Dose rate, millirem/year
Thyroid	3.3×10^{-3}
Lungs	6.9×10^{-6}
Total body	1.8×10^{-6}
Ovaries	1.2×10^{-7}
Total endosteal cells	6.4×10^{-7}
Testes	6.4×10^{-8}

Note that inhalation calculations are not made for ^{85}Kr since it is assumed that the inhalation doses from noble gases are insignificant when compared to air immersion doses.

B.2.2.2 <u>Ingestion doses</u>. The calculation of dose to humans as a result of ingestion of contaminated food is analagous to the calculation of inhalation doses as expressed by Eq. (B.16) (ref. 7),

$$R_{ij}^f = C_i U_i^f D_{ij}^f \qquad (B.17)$$

where

R_{ij}^f = dose rate to organ j as a result of eating food contaminated with radionuclide i (rem/year),

C_i = concentration of radionuclide i in the food of interest (μCi/kg or μCi/liter),

U_i^f = rate of intake of the food of interest containing radio nuclide i (kg/year or liter/year),

D_{ij}^f = dose conversion factor for radionuclide i and organ j due to ingestion of contaminated food (rem/μCi intake).

B.2.2.2.1 <u>Ingestion of fresh produce</u>. The concentration of ^{131}I in leafy vegetables from the ground-level release of $^{131}I_2$ being considered in this appendix was found in Sect. B.1.1.3.2 to be

$$C_i = C_i^v = 2.3 \times 10^{-1} \text{ pCi/kg}$$
$$= 2.3 \times 10^{-7} \text{ μCi/kg}.$$

Table 6.1 gives an annual average adult intake rate for leafy vegetables of

$$U_i^f = 18 \text{ kg/year },$$

assuming all of the leafy vegetables a person consumes are grown in the area effected by the release being considered. From Table 7.1 the ingestion dose conversion for $i = ^{131}I$ and $j = $ thyroid is found to be

$$D_{ij}^{f} = 1.81 \text{ rem/}\mu Ci \quad .$$

It follows from Eq. (B.17) that the dose rate to the thyroid in this example is

$$R_{ij}^{f} = (2.3 \times 10^{-7} \mu Ci/kg)(18 \frac{kg}{year})(1.81 \frac{rem}{\mu Ci})$$
$$= 7.5 \times 10^{-6} \text{ rem/year}$$
$$= 7.5 \times 10^{-3} \text{ millirem/year.}$$

The dose rates to other organs from the ingestion of ^{131}I in leafy vegetables are also calculated from Eq. (B.17):

Organ	Dose rate, millirem/year
Thyroid	7.5×10^{-3}
Lungs	1.2×10^{-6}
Total body	3.7×10^{-6}
Ovaries	2.5×10^{-7}
Total endosteal cells	1.4×10^{-6}
Testes	1.5×10^{-7}

B.2.2.2.2 <u>Ingestion of milk</u>. From Sect. B.1.1.3.3 the concentration of ^{131}I in milk for the release being considered here is

$$C_i = C_i^{m} = 1.6 \text{ pCi/liter}$$
$$= 1.6 \times 10^{-6} \mu Ci/liter \quad .$$

Once again assuming all of the milk a person consumes comes from the area surrounding the release being considered, Table 6.1 gives an annual average adult intake rate for milk

$$U_i^f = 112 \text{ liter/year.}$$

The appropriate dose conversion factors may again be found in Table 7.1. For $i = {}^{131}I$ and $j = $ thyroid, it is again found that

$$D_{ij}^f = 1.81 \text{ rem/}\mu Ci \quad .$$

Applying Eq. (B.17), the dose rate to the thyroid is

$$R_{ij}^f = (1.6 \times 10^{-6} \text{ }\mu Ci/l)(112 \text{ liter/year})(1.81 \text{ rem/}\mu Ci)$$

$$= 3.2 \times 10^{-4} \text{ rem/year}$$

$$= 3.2 \times 10^{-1} \text{ millirem/year.}$$

Similar calculations for the dose rate to other organs due to the ingestion of ${}^{131}I$ in milk are included in the following:

Organ	Dose rate, millirem/year
Thyroid	3.2×10^{-1}
Lungs	5.1×10^{-5}
Total body	1.6×10^{-4}
Ovaries	1.1×10^{-5}
Total endosteal cells	6.0×10^{-5}
Testes	6.2×10^{-6}

B.2.2.2.3 Ingestion of meat. The dose to various organs as a result of the ingestion of ${}^{131}I$ in meat may also be found using Eq. (B.17) and the following input parameters:

$$C_i = C_i^F = 2.9 \times 10^{-1} \text{ pCi/kg (Sect. B.1.1.3.4),}$$

$$= 2.9 \times 10^{-7} \text{ }\mu Ci/kg$$

U_i^f = 32 kg/year (Table 6.1),

D_{ij}^f = values for various organs found in Table 7.1.

If we again assume that the person consumes only beef raised in the area influenced by the release under consideration, the dose rates that result are as follows:

Organ	Dose rate, rem/year
Lungs	2.6×10^{-7}
Total body	8.2×10^{-9}
Ovaries	5.7×10^{-10}
Total endosteal cells	3.1×10^{-9}
Testes	3.3×10^{-10}
Thyroid	1.7×10^{-5}

B.2.2.2.4 <u>Ingestion of fish</u>. Equation (B.17) may also be used to calculate the dose due to the ingestion of fish containing ^{137}Cs as calculated in Sect. B.1.2.2. From that section,

$$C_i = CF_i = 1.7 \times 10^{-9} \text{ Ci/kg}$$
$$= 1.7 \times 10^{-3} \text{ } \mu\text{Ci/kg} \text{ .}$$

Table 6.1 gives, for freshwater finfish,

$$U_i^f = 4.4 \text{ kg/year .}$$

The dose conversion factors for i = ^{137}Cs are also found in Table 7.1. For j = total body,

$$D_{ij}^f = 4.91 \times 10^{-2} \text{ .}$$

Assuming all of the fish consumed by this person contains C_i, the resulting annual dose rate to the total body is

$$R_{ij}^f = (1.7 \times 10^{-3} \frac{\mu Ci}{kg})(4.4 \frac{kg}{year})(4.91 \times 10^{-2} \frac{rem}{\mu Ci})$$

$$= 3.7 \times 10^{-4} \text{ rem/year}$$

$$= 3.7 \times 10^{-1} \text{ millirem/year.}$$

Similar calculations give the additional results included below for the other organs listed in Table 7.1:

Organ	Dose rate, millirem/year
Total body	3.7×10^{-1}
Lungs	1.5×10^{-1}
Ovaries	5.6×10^{-1}
Total endosteal cells	6.0×10^{-1}
Testes	5.0×10^{-1}

B.2.3 <u>Tritium and Carbon-14 Dose Calculations</u>

B.2.3.1 <u>Tritium doses</u>. Assume ^3H is being released from an elevated source under the meteorological and location conditions considered for the ^{85}Kr release in Sect. B.1.1.2. From this section,

$$\frac{\chi}{Q} = 1.0 \times 10^{-6} \text{ s/m}^3 \quad .$$

If we assume for this ^3H release that $Q = 1 \times 10^{-6}$ Ci/s, then the air concentration is

$$\chi = 1.0 \times 10^{-12} \text{ Ci/m}^3$$

$$= 1.0 \times 10^{-6} \text{ pCi/cm}^3 \quad .$$

The total ingestion dose from ^3H for food and water can be calculated using Eq. (8.1),

$$D_t = C_f \chi + C_w \chi \ . \tag{B.18}$$

Using the values of C_f and C_w found in Sect. 8.1.2, the ^{3}H ingestion dose for the release considered here is

$$D_t = (6.18 \ \frac{rem\text{-}cm^3}{pCi\text{-}year}) \ (1.0 \times 10^{-6} \ \frac{pCi}{cm^3})$$

$$+ \ (5.70 \ \frac{rem\text{-}cm^3}{pCi\text{-}year}) \ (1.0 \times 10^{-6} \ \frac{pCi}{cm^3})$$

$$= 1.2 \times 10^{-5} \ rem/year$$

$$= 1.2 \times 10^{-2} \ millirem/year.$$

The dose due to the inhalation and skin absorption of ^{3}H is found from Eq. (B.16). For the total endosteal cells,

$$\chi_i^a = 1.0 \times 10^{-6} \ \mu Ci/m^3,$$

$$B_r = 8.03 \times 10^3 \ m^3/year \ (Table \ 6.1),$$

$$D_{ij}^{br} = 9.85 \times 10^{-5} \ rem/\mu Ci \ (Table \ 7.1),$$

and

$$R_{ij}^{br} = (1.0 \times 10^{-6} \ \frac{\mu Ci}{m^3})(8.03 \times 10^3 \ \frac{m^3}{year})(9.85 \times 10^{-5} \ \frac{rem}{\mu Ci})$$

$$= 7.9 \times 10^{-7} \ rem/year$$

$$= 7.9 \times 10^{-4} \ millirem/year.$$

For all other organs listed in Table 7.1, $D_{ij}^{br} = 1.25 \times 10^{-4}$ resulting in $R_{ij}^{br} = 1.0 \times 10^{-3}$ millirem/year.

B.2.3.2 <u>Carbon-14 doses.</u> Assume that $^{14}CO_2$ is released with the ^{3}H considered in Sect. B.2.3.1. As a result,

$$\frac{\chi}{Q} = 1.0 \times 10^{-6} \text{ s/m}^3$$

for this release situation also. If $Q = 5.0 \times 10^{-7}$ Ci/s,

$$\chi = 5.0 \times 10^{-13} \text{ Ci/m}^3$$
$$= 5.0 \times 10^{-7} \text{ pCi/cm}^3$$

According to Sect. 8.2 the primary dose to man from ^{14}C is via the ingestion pathway. This dose rate for various organs is equal to the product of the concentration of ^{14}C in air and the appropriate dose conversion factor listed in Table 8.1. The resulting dose rates from the air concentration noted above are as follows:

Organ	Dose rate, millirem/year
Whole body	6.0×10^{-1}
Red marrow	1.0
Lungs	2.5×10^{-1}
Endosteal cells	9.0×10^{-1}
Stomach wall	3.8×10^{-1}
Lower large intestine wall	4.6×10^{-1}
Thyroid	$2.7 \times 10{-1}$
Liver	3.6×10^{-1}
Kidneys	3.3×10^{-1}
Testes	2.2×10^{-1}
Ovaries	2.2×10^{-1}

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132. R. E. Cooper, Savannah River Laboratory, Aiken, South Carolina 29801
133. J. C. Corey, Savannah River Laboratory, Aiken, South Carolina 29801
134. T. V. Crawford, Savannah River Laboratory, Aiken, South Carolina 29801
135. R. G. Cuddihy, Inhalation Toxicology Research Institute, 5200 Gibson Blvd., S.E., Albuquerque, New Mexico 87108
136. Walter M. Culkowski, National Oceanic and Atmospheric Administration, 106 Greystone Drive, Oak Ridge, Tennessee 37830
137. F. L. Culler, Electric Power Research Institute, P.O. Box 10412, Palo Alto, California 94303
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141. P. B. Dunaway, Office of Effects Evaluation, Nevada Operations Office, Department of Energy, Las Vegas, Nevada 87114

142. D. N. Edgington, Ecology Section, Radiological and Environmental Research Division, Argonne National Laboratory, 9700 S. Cass Avenue, Argonne, Illinois 60439

143. G. G. Eichholz, School of Nuclear Engineering, Georgia Institute of Technology, Atlanta, Georgia 20332

144. W. H. Ellett, CSD, Office of Radiation Programs, U.S. Environmental Protection Agency, Washington, D.C. 20460

145. W. H. Espey, Jr., Espey Houston Association, Inc., 3010 South Lamar, Austin, Texas 78704

146. H. L. Falk, National Institute of Environmental Health Sciences, P.O. Box 12233, Research Triangle Park, North Carolina 27709

147. J. E. Fitzgerald, Criteria and Standards Division (AW-460), Office of Radiation Programs, U.S. Environmental Protection Agency, 401 M Street, S.W., Washington, D.C. 20460

148. Edward H. Fleming, University of California, Lawrence Livermore Laboratory, Mail Stop L-1, Box 808, Livermore, California 94550

149. Ted Fowler, U.S. Environmental Protection Agency, Eastern Environmental Radiation Facility, P.O. Box 3009, Montgomery, Alabama 36109

150. Mahlon Gates, Nevada Operations Office, Department of Energy, P.O. Box 1676, Las Vegas, Nevada 99101

151. F. A. Gifford, Jr., Air Resources Atmospheric Turbulence and Diffusion Laboratory, National Oceanic and Atmospheric Administration, Oak Ridge, Tennessee 37830

152. Earnest F. Gloyna, University of Texas, College of Engineering, Cockell Hall, Austin, Texas 78712

153. Carl V. Gogolak, Department of Energy, Environmental Measurement Laboratory, 376 Hudson Street, New York, New York 10014

154. John Golden, Commonwealth Edison Company, P.O. Box 767, Chicago, Illinois 60690

155. Abraham S. Goldin, Office of Radiation Programs, U.S. Environmental Protection Agency, Washington, D.C. 20460

156. Marvin Goldman, University of California, Davis, California 95616

157. R. L. Gotchy, Radiological Assessments Branch, Nuclear Regulatory Regulation, U.S. Nuclear Regulatory Commission, Washington, D.C. 20555

158. Monica Gromms, Allied General Nuclear Services, P.O. Box 847, Barnwell, South Carolina 29812

159. Wayne C. Hanson, Environmental Studies, University of California, Los Alamos Scientific Laboratory, p.O. Box 1663, Los Alamos, New Mexico 97544

160. D. F. Harmon, Office of Regulation, U.S. Nuclear Regulatory Commission, Washington, D.C. 20555

161. Florence L. Harrison, University of California, Lawrence Livermore Laboratory, P.O. Box 808, Livermore, California 94550

162. Monte Hawkins, Safety and Environmental Control, Allied General Nuclear Service, P.O. Box 847, Barnwell, South Carolina 29812

163. David Hayes, E.I. duPont, Savannah River Laboratory, SRL-773-16A, Aiken, South Carolina 29801

164. J. W. Healy, Health Physics Division, Los Alamos Scientific Laboratory, P.O. Box 1663, Los Alamos, New Mexico 97544

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168. S. L. Iverson, Whiteshell Nuclear Research Est., Pinawa, Manitoba ROE 1LO, Canada

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170. Stephen Jinks, Institute of Environmental Medicine, New York University Medical Center, Long Meadow Road, Tuxedo, New York 10987

171. B. Kahn, Interdisciplinary Sciences, Georgia Institute of Technology, Atlanta, Georgia 30332

172. Jacob Kastner, U.S. Nuclear Regulatory Commission, Washington, D.C. 20555

173. M. D. Kelley, General Electric Company, 175 Curtner Avenue, San Jose, California 95125

174. J. Kline, Radiological Physics Division Ecological Studies, Argonne National Laboratory, 9700 South Cass Avenue, Argonne, Illinois 60439

175. E. C. Lazer, Standards and Regulation Evaluation Division, U.S. Environmental Protection Agency, 401 M Street, S.W., Washington, D.C. 20460

176. Joan Lefler, TS-792, Environmental Review Division, Office of Testing and Evaluation, Office of Toxic Substances, U.S. Environmental Protection Agency, Rm. 623 East Tower, 401 M Street S.W., Washington, D.C. 20460

177. Carl L. Lindeken, Lawrence Livermore Laboratory, P.O. Box 808, Livermore, California 94550

178. A. H. Lu, Division of Laboratories and Research, New York State Department of Health, New Scotland Avenue, Albany, New York 12201

179. Paul J. Magno, Office of Radiation Programs, U.S. Environmental Protection Agency, Washington, D.C. 20460

180. J. R. Maher, Division of Technology Overview, Department of Energy, Washington, D.C. 20545

181. Bruce J. Mann, Evaluation Branch, U.S. Environmental Protection Agency, Office of Radiation Programs, P.O. Box 15027, Las Vegas, Nevada 89114

182. Earl H. Markee, Jr., U.S. Nuclear Regulatory Commission, Mail Stop P-214, Washington, D.C. 20555

183. Dan E. Martin, U.S. Nuclear Regulatory Commission, Washington, D.C. 20555

184. P. J. Mellinger, Exxon Nuclear Company, 2955 George Washington Way, Richland, Washington 99352

185. W. A. Mills, Criteria and Standards Division, U.S. Environmental Protection Agency, 401 M Street, S.W., Washington, D.C. 20460

186. P. K. Misra, Technology Development and Appraisal Section, Air Resources Branch, Ontario Ministry of the Environment, 880 Bay Street, 4th Floor, Toronto, Ontario, M5S 1Z8, Canada

187. Harold J. Monroe, Tennessee Valley Authority, Rm. 272, 401 Building, Chattanooga, Tennessee 37401

188. H. A. Morewitz, Atomics International Division, Rockwell International, 8900 De Soto Avenue, Canoga Park, California 91304

189. K. Z. Morgan, School of Nuclear Engineering, Georgia Institute of Technology, Atlanta, Georgia 30332

190. R. B. Neel, U.S. Nuclear Regulatory Commission, Washington, D.C. 20555

191. Christopher B. Nelson, Office of Radiation Programs, U.S. Environmental Protection Agency, Washington, D.C. 20460

192. Yook C. Ng, Biomedical and Environmental Research Division, Lawrence Livermore Laboratory, P. O. Box 808, Livermore, California 94550

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200. John M. Palms, Emory University, Atlanta, Georgia 30322

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202. Robert G. Patzer, U.S. Environmental Protection agency, P.O. Box 15027, Las Vegas, Nevada 89114

203. Al Payne, University of North Carolina, Chapel Hill, 3811 Cobb Street, Garner, North Carolina 27529

204. H. R. Payne, U.S. Environmental Protection Agency, Region IV, 1421 Peachtree Street, Atlanta, Georgia 30309

205. M. M. Pendergast, Savannah River Laboratory, Aiken, South Carolina 29801

206. Harold Peterson, Battelle Memorial Institute, 505 King Avenue, Columbus, Ohio 43201

207. Warren Piver, National Institute of Environmental Health Sciences, P.O. Box 12233, Research Triangle Park, North Carolina 27709

208. J. W. Poston, School of Nuclear Engineering, Georgia Institute of Technology, Atlanta, Georgia 30332

209. O. G. Raabe, Radiobiology Laboratory, University of California at Davis, Davis, California 95616

210. D. P. Rall, National Institute of Environmental Health Sciences, P.O. Box 12233, Research Triangle Park, North Carolina 27709

211. Gerald Rausa, Office of Research, U.S. Environmental Protection Agency, Washington, D.C. 20460

212. George W. Reynolds, Tennessee Valley Authority, Rt. 7, Box 402, Florence, Alabama 35630

213. Theodore R. Rice, National Marine Fisheries Service, Beaufort Laboratory, Beaufort, North Carolina 28516
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215. A. P. Roeh, HTGR Fuel Reprocessing Program, Allied Chemical Corporation, 550 Second Street, Idaho Falls, Idaho 83401
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221. R. G. Schreckhise, Battelle-Pacific Northwest Laboratory, P.O. Box 999, Richland, Washington 99352
222. David L. Schreiber, U.S. Nuclear Regulatory Commission, Mail Stop P-214, Washington, D.C. 20555
223. Norman Schwartz, Dames and Moore Incorporated, 7101 Wisconsin Avenue, Washington, D.C. 20014
224. G. A. Sehmel, Division of Biology and Medicine, Battelle Pacific Northwest Laboratories, Richland, Washington 99352
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229. Bruce Smith, 12917 Penrose Street, Rockville, Maryland 20853
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233. J. K. Soldat, Pacific Northwest Laboratories, Battelle Memorial Institute, Richland, Washington 99352
234. G. E. Start, National Oceanic and Atmospheric Administration, 550 Second Street, Idaho Falls, Idaho 93401
235. Keith G. Steyer, Office of Standards Development, U.S. Nuclear Regulatory Commission, Washington, D.C. 20555
236. John A. Strand, Battelle-Pacific Northwest Laboratories, P.O. Box 999, Richland, Washington 99352
237. Dennis Strenge, Battelle-Pacific Northwest Laboratories, P.O. Box 999, Richland, Washington 99352
238. Frank J. Swanburg, U.S. Nuclear Regulatory Commission, Washington, D.C. 20555
239. Jerry Swift, Technology and Assessments, Office of Radiation Programs, U.S. Environmental Protection Agency, 401 M Street, S.W., Washington, D.C. 20460

240. W. L. Templeton, Ecosystems Department, Battelle-Pacific Northwest Laboratories, Richland, Washington 99352

241. Joseph D. Teresi, Breeder Reactor Department, General Electric Company, 210 Dequire, Sunnyvale, California 94086

242. S. E. Thompson, Lawrence Livermore Laboratory, University of California, Livermore, California 94550

243. A. N. Tse, Product Standards Branch, Directorate of Regulatory Standards, U.S. Nuclear Regulatory Commission, Washington, D.C. 20555

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245. I. Van der Hoven, National Oceanic and Atmospheric Administration, 8060 13th Street, Silver Spring, Maryland 20910

246. H. A. Vanderploeg, National Oceanic and Atmospheric Administration, Great Lakes Environmental Research Laboratory, 2300 Washtenaw Avenue, Ann Arbor, Michigan 48104

247. E. C. Watson, Battelle-Pacific Northwest Laboratories, P.O. Box 999, Richland, Washington 99352

248. J. E. Watson, Jr., School of Public Health, University of North Carolina, Chapel Hill, North Carolina 27514

249. R. Watters, Division of Biomedical and Environmental Research, Department of Energy, Washington, D.C. 20545

250. James R. Watts, Savannah River Laboratory, Aiken, South Carolina 29801

251. Paul Webb, General Electric Company, 175 Curtner Avenue, San Jose, California 95125

252. Gary C. White, Los Alamos Sandia Laboratory, 3303 B Orange, Los Alamos, New Mexico 87544

253. B. R. Wheeler, HTGR Program Office, Allied Chemical Corporation, 550 Second Street, Idaho Falls, Idaho 83401

254. F. W. Whicker, Department of Radiology and Radiation Biology, Colorado State University, Fort Collins, Colorado 80521

255. W. H. Wilkie, Tennessee Valley Authority, River Oaks Building, Muscle Shoals, Alabama 35660

256. W. R. Woodall, Jr., Georgia Power and Light Company, Atlanta, Georgia 30303

257. K. Y. Wong, Ontario Hydro, Physics Department, Pickering Generating Station, P.O. Box 160, Pickering, Ontario, Canada

258. R. W. Wood, Division of Biology and Medicine, Department of Energy, Washington, D.C. 20545

259. S. Yaniv, Office of Nuclear Regulatory Research, Mail Stop A1-3002, U.S. Nuclear Regulatory Commission, Washington, D.C. 20555

260. Claude Yarbro, Research and Technical Support Division, DOE-ORO, Oak Ridge, Tennessee 37830

261. Nobuhiro Yotsukura, U.S. Geological Survey, Water Resources Division National Center, MS 430, 12201 Sunrise Valley Drive, Reston, Virginia 22092

262. Paul L. Ziemer, Bionucleonics Department, Pharmacy Building, Purdue University, West Lafayette, Indiana 47907

263. Office of Assistant Manager, Energy Research and Development, Department of Energy, Oak Ridge Operations Office, Oak Ridge, Tennessee 37830

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